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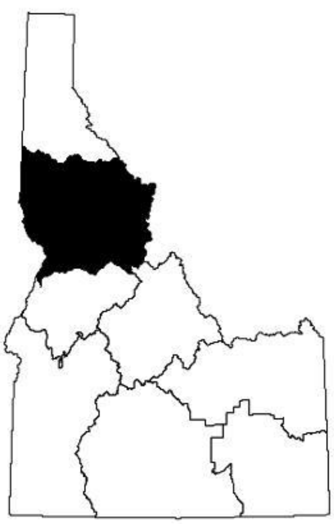

CLEARWATER REGION
2018

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DEER CREEK RESERVOIR: TIGER TROUT AND RAINBOW TROUT EVALUATION

ABSTRACT

Surveys were conducted in Deer Creek Reservoir to evaluate changes in Golden Shiner *Notemigonus crysoleucas* (GS) abundance and length due to stocking catchable-size tiger trout (Brown Trout *Salmo trutta* X Brook Trout *Salvelinus fontinalis*; TT), and evaluate whether abundance and length of stocked hatchery trout were adequate to meet fishery objectives. We also collected otoliths from TT to evaluate age, growth, and mortality, analyzed diet composition for all trout species, and tagged catchable-size Rainbow Trout *Oncorhynchus mykiss* (RBT) and TT to evaluate angler exploitation. Surveys were conducted in 2018 using boat-mounted electrofishing of historic transects and gill nets, and compared to findings from previous years. Our results indicate that GS catch-per-unit-effort (CPUE; fish/h) has remained stable, while mean total length has increased since 2014. Diet analysis shows that GS were present in 28% of TT stomachs, while no trout of any species prey on GS until trout reach ~240 mm in length. There was no significant change in mean relative weight of TT for fall electrofishing surveys from 2016 to 2018, while mean relative weight of TT > 350 mm was significantly higher than for TT < 350 mm. Annual mortality of TT was estimated at 54%, while annual growth of TT was ~50 mm. Angler exploitation of TT was 14% and total use was 15% through 730 days-at-large, similar to previous years. CPUE of RBT was significantly lower in 2018, although mean length was longer. Angler exploitation of RBT was 42%, significantly higher than previous surveys. Brook Trout CPUE continued to decline, while no WCT have been sampled since 2016. Our analysis suggests that the GS population appears to be stabilizing, indicating that TT are having little impact, or we are unable to detect changes based on our sampling. Diet analysis indicated that TT are successfully preying upon GS, but this does not appear to be affecting GS abundance. With no change in TT relative weight over time, our data indicates that DCR is not being overstocked. Additionally, the higher relative weights for larger TT suggests they are feeding successfully and can potentially provide the put-grow-take fishery that was the intent of this stocking program. Annual mortality of TT was lower than estimated for hatchery RBT in Idaho reservoirs. However, due to our limited sample size, we recommend collecting more data to improve our analysis of age, growth, and mortality. The angler total use rate for TT has not increased significantly for fish stocked from 2016 to 2018. Rainbow Trout exploitation exceeded our management goal; however, due to the significant increase compared to previous surveys, we recommend an additional survey. Low catch of Brook Trout with multiple gear types confirms that there is not a gear bias for this species with electrofishing. It is questionable whether continued stocking of BKT or WCT would provide sufficient opportunity. We do not recommend further stocking of WCT. At this time, an additional year of sampling for GS and hatchery trout (including the collection of otoliths from TT) is needed to finalize our evaluation and determine if changes to our stocking rates, sizes, and fishing regulations should be considered.

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INTRODUCTION

Deer Creek Reservoir (DCR) was constructed during 2003 by impounding Deer Creek, a tributary of Reeds Creek that flows into Dworshak Reservoir. Deer Creek Reservoir is an important part of the region's lowland lake program as it allows trout harvest in an area where all stream fishing is under restrictive harvest regulations (two trout per day). It also adds diversity to our fisheries program as it is the only lowland lake managed solely as a cold-water fishery. It was originally managed with put-and-take Rainbow Trout *Oncorhynchus mykiss* (RBT) and put-grow-take Westslope Cutthroat Trout *O. clarkii lewisi* (WCT) fisheries. Sterile Brook Trout *Salvelinus fontinalis (BKT)* were stocked beginning in 2012 as an additional put-grow-take fishery. Deer Creek Reservoir was managed with general trout regulations until 2019, when it was changed to six trout, only two of which may be tiger trout (none less than 356 mm).

Golden Shiner *Notemigonus crysoleucas* (GS) appeared in DCR soon after the reservoir filled. Upon their discovery in 2006 and again in 2010, the reservoir was treated with rotenone in attempts to eradicate this invasive species (Hand 2010; Hand et. al. 2013). The attempts to eradicate GS from DCR were based on several risks: 1) GS are effective planktivores, and an overabundance of GS in DCR could potentially reduce the quality and quantity of the zooplankton available for trout; and 2) we were concerned that GS might spread downstream into Dworshak Reservoir which supports an important kokanee fishery that has been found to generate more than $4 million in fishing-related expenditures annually (IDFG, *unpublished data*).

Unfortunately, GS were again discovered in DCR in 2012. The failure of the rotenone treatments was likely due to a combination of factors including the high level of habitat complexity within the reservoir (large slash piles had been left to provide habitat), springs or seeps that could provide clean water refuge, and GS resistance to rotenone. Golden Shiner have a natural resistance to rotenone and are capable of developing a higher resistance to rotenone which would increase each time the same population is exposed (Orciari 1979). If the initial renovation was not 100% effective, any surviving GS would have the potential of creating a rotenone-resistant population. Additionally, GS were found in several ponds in the nearby Schmidt Creek drainage (near Weippe, Idaho) during the construction of Deyo Reservoir. Nez Perce Tribe fisheries biologists also reported finding GS in nearby drainages including Orofino Creek and Jim Ford Creek. This indicated that GS were widespread, making complete eradication nearly impossible, and making it highly likely that GS had already reached Dworshak Reservoir.

With the realization that rotenone treatments were not effective at eliminating GS, we researched different trout species that could prey upon Golden Shiner, provide desirable fishing opportunities, and not pose a risk to downstream fisheries. Ultimately, we decided to introduce tiger trout (TT; Brown Trout *Salmo trutta* X Brook Trout) into DCR as they had been reported to be an effective predator on minnow species (Sheerer et. al. 1987; Winters 2014), provide desirable fisheries (Winters 2014), and are sterile, posing no little risk to downstream fisheries. Our hope was that TT would utilize GS as a prey source, thus improving the food base for trout that depend on zooplankton, and provide a unique fishing opportunity in the region. With this in mind, we began stocking fingerling TT (50 - 75 mm) in DCR in the spring of 2014.

Surveys conducted in 2014 confirmed our concern regarding zooplankton density and quality (i.e. size), as sampling revealed a decline in zooplankton size and density compared to previous years before GS were present (Hand et al. 2017). This decline in food resources may have been a primary reason why only one TT was sampled in 2014 and 2015, and would likely result in future poor growth and survival of trout dependent on zooplankton. Golden Shiner were only present in the stomach contents of Rainbow Trout and Brook Trout > 250 mm. The apparent lack of success in establishing a TT fishery through the fingerling stockings suggested that changes to our stocking strategy were necessary. Decreasing or eliminating the stocking of fingerling trout, and stocking larger trout (TT, RBT, or BKT > 250 mm) could increase their likelihood of survival, decrease the predation pressure on zooplankton, and increase predation of GS. Thus, we changed our strategy to stock approximately 2,500 "catchable-size" TT (170 - 360 mm) annually. The first catchable TT were stocked in 2016, and sampling occurred annually from 2016 to 2018 to evaluate the success of this new strategy.

OBJECTIVES

1. Evaluate whether abundance and length of Golden Shiner in Deer Creek Reservoir have changed after stocking catchable-size tiger trout.

2. Assess whether abundance and sizes of tiger trout, Rainbow Trout, and Brook Trout being stocked into Deer Creek Reservoir are adequate to meet fishery needs.

STUDY AREA

Deer Creek Reservoir is located in Clearwater County, Idaho, 21 km north of Pierce, Idaho (Figure 1). It is a 47.0-ha reservoir located at an elevation of 1,006 m. It has a maximum depth of 11 m, and a maximum volume of 936,000 m^3. All land in the DCR watershed is owned by PotlatchDeltic and is used primarily for timber harvest. Idaho Department of Fish and Game leases the reservoir property. Today, the reservoir is managed to provide family fishing opportunities for cold-water species.

METHODS

FIELD SAMPLING

Golden Shiner

The GS population in DCR was sampled through boat electrofishing surveys conducted on July 28 and August 20, 2018. Electrical waveform consisted of pulsed DC produced by a Honda EU7000iAT1 generator and a Midwest Lakes Electrofishing Systems (MLES) Infinity pulsator. In order to maintain consistency, we conducted these surveys at the same time of year, and using the same methods as in 2014 and 2017. Additionally, we sampled the same ten, 50-m transects that have been electrofished in past GS sampling efforts (Figure 2). These transects were originally selected from random shoreline GPS points developed during a vegetation survey conducted in 2012 (Hand et. al. 2016). All transects were sampled during each survey with the boat moving along the shoreline in a clockwise direction. The surveys were conducted at night, and we attempted to net all fish observed. Golden Shiner were measured for total length (TL; mm); weights were not recorded.

Trout

The trout in DCR were surveyed three times in 2018 using different gear types (2 electrofishing samples, 1 gill net sample) to achieve our objectives. An electrofishing survey was conducted for trout on October 23, 2018 to compare with previous population surveys conducted in fall of 2016 and 2017. The equipment used is described in the Golden Shiner sampling methodology. In order to maintain sampling consistency, we utilized the same transects that have been used for all trout sampling conducted in DCR (Figure 3). This sampling consisted of one hour of electrofishing, divided into six, 10-minute transects, with fish collected in each transect processed and recorded separately. This allowed a variance to be calculated around the mean CPUE, allowing statistical comparisons to the surveys conducted in 2016 and 2017. Electrofishing was conducted along the shoreline in a clockwise direction. The survey was conducted at night, and we attempted to net all trout observed. Trout were identified by species, with TL (mm) and weight (g) recorded for each fish.

An electrofishing survey was conducted on June 6, 2018 specifically to collect TT otoliths. We collected otoliths at this time, as opposed to utilizing the fall electrofishing survey, to collect samples prior to the TT stocking scheduled for June 7, 2018. Boat-mounted electrofishing was conducted using the equipment described in the Golden Shiner sampling methodology. The equipment and protocol utilized were as described above for the fall population electrofishing survey, except only TT were captured. Otoliths were collected using wire cutters to open the fish, and tweezers to locate and remove the otoliths. Each set of otoliths was stored in a coin envelope labelled with the date, reservoir name, fish species, TL, and weight. We also evaluated stomach contents of the sacrificed TT to increase our sample size. This analysis was conducted on the reservoir at the same time otoliths were removed. Stomach contents were separated into five different categories: empty, zooplankton, GS, insects, and detritus (no other items were observed). Accurate lengths of GS could not be determined due to digestion; therefore, all items were recorded as presence/absence.

The trout stocked in DCR were surveyed through a gill-net survey conducted on October 23, 2018 to increase our sample size for diet analysis. We utilized gill nets in 2018 to maintain consistency with previous diet analysis surveys conducted from 2014 to 2017. The presence of large quantities of wood throughout the reservoir, especially the upper 1/3, prevent us from placing nets in DCR randomly. Therefore, gill net sites were spread around the reservoir as much as possible, while being set in areas least likely to be damaged (Figure 4). The survey consisted of four gill nets set for 15.0 hours overnight. Floating monofilament experimental gill nets 36 m long and 1.8 m high were used. The nets were divided into six equal size panels with bar mesh sizes of 10.0, 12.5, 18.5, 25.0, 33.0 and 38.0 mm. Monofilament diameter ranged from 0.15 to 0.20 mm. Nets were set perpendicular to the shoreline and anchored in place to prevent the net from drifting. The smallest mesh end was tied to shore, and the largest mesh end towards the middle of the reservoir. Weight (g) and TL (mm) were recorded for all trout collected, and they were dissected to identify stomach contents.

To evaluate angler exploitation of TT and RBT, a subsample of hatchery catchable-size RBT (*n* = 200) stocked on May 18, 2018 and TT (*n* = 200) stocked on June 7, 2018 were tagged at DCR prior to stocking. These fish were randomly selected by netting fish directly from the hatchery truck into a holding tank. All fish selected were tagged with Hallprint model FD-94 anchor tags. Each fish tagged was measured for TL (mm) and weight (g). Tagging data (date, location, species, TL, weight, tag number) was submitted to the IDFG Nampa Research Office and uploaded to the IDFG “Tag You’re It” database.

DATA ANALYSIS

Golden Shiner

To evaluate whether the abundance of GS differed between years (2014 to 2018), we compared mean CPUE (fish/transect) using a single-factor ANOVA with a significance level of $\alpha = 0.10$. If ANOVA showed that a significant difference occurred, we used a Tukey-Kramer Test to determine which years significantly differed from one another with a significance level of $\alpha = 0.10$. For each year, we pooled July and August surveys to increase sample size and reduce variability.

To evaluate whether the size distribution of GS differed between years (2014 to 2018), we compared the mean TL using a single-factor ANOVA with a significance level of $\alpha = 0.10$. If ANOVA showed that a significant difference occurred, we used a Tukey-Kramer Test to determine which years significantly differed from one another with a significance level of $\alpha = 0.10$. For each year, we pooled July and August surveys to increase sample size and reduce variability.

Trout

To evaluate whether the abundance and sizes of TT, RBT, and BT stocked into DCR were adequate to meet fishery needs, we assessed various population dynamic attributes on this fishery including recruitment (stocking rate), age, growth, relative abundance, and mortality.

Mean CPUE (fish/h) from the fall electrofishing survey was calculated for TT, RBT, BKT, and WCT and compared with data collected in 2016 and 2017. This data was not compared to trout data collected in 2014 or 2015 because these surveys were conducted at a different time of year (i.e. fall vs. spring) or with different gear (gill net). We used a single-factor ANOVA with a significance level of $\alpha = 0.10$ to evaluate whether mean CPUE and mean TL differed among years (2016 - 2018). If ANOVA showed that a significant difference occurred, we used a Tukey-Kramer Test to determine which years significantly differed from one another with a significance level of $\alpha = 0.10$.

Relative weight (W_r; Wege and Anderson 1978; Neumann et al. 2012) was calculated for each TT sampled in November by electrofishing. The relative weight equation is:

$$W_r = \frac{W}{W_s} * 100$$

where W is the observed weight of the fish and W_s is the length-specific standard weight predicted by a weight-length regression. This equation is:

$$log_{10}\ Ws = a + (b * log_{10}\, total\ length)$$

where *a* is the intercept and *b* is the slope of standard weight equations developed for many fish species (Wege and Anderson 1978; Neumann et al. 2012). We used the length-weight relationship for Brown Trout ($a = -4.867$ and $b = 2.96$) from Blackwell et al. (2000) and used by Messner and Schoby (2019) for tiger trout evaluations in Wallace Lake, Idaho. Significant difference in mean relative weight among TT (all TT and just those individuals > 350 mm) surveyed during fall electrofishing (2016 - 2018) was evaluated using a single-factor ANOVA with a significance level of $\alpha = 0.10$. If ANOVA showed that a significant difference occurred, we used a Tukey-Kramer Test to determine which years significantly differed from one another with a significance level of $\alpha = 0.10$. We also assessed differences in body condition for TT < 350 mm compared to those > 350 mm for fish collected each year from 2016 to 2018 using two-tailed *t*-tests (assuming equal variance) with a significance level of $\alpha = 0.10$.

Otoliths were analyzed to evaluate TT age, growth, and mortality, and their effects on the fishery. Otoliths were mounted in epoxy resin and sectioned with a Bueler Isomet low speed saw. Sections were photographed using a Leica M80 stereoscope, with a TL5000 Ergo transmitted light base, and an IC80HD camera. The software ImageJ was utilized to mark annuli and measure distance. Age was estimated by counting annuli. Back-calculation of lengths at age were determined using the Fraser-Lee equation (Quist et al. 2012):

$$L_i = c + (L_c - c)(S_i/S_c)$$

Where:

- c = size of each fish at time of scale formation
- S_i = the scale radius at annulus formation
- S_c = the overall scale radius
- L_i = the length at annulus formation
- L_c = the fish length at capture

The variable "*c*" was determined by the Y-intercept of a regression line plotting scale radius (x-axis) versus fish length (y-axis).

Mortality rates were calculated by developing a catch curve for TT (Miranda and Bettoli 2007; Allen and Hightower 2010). The catch curve plots the age of the fish collected versus the $\log^e$ of the number of fish captured. The slope of this line represents the annual instantaneous mortality rate (Z). The annual survival rate (S) is calculated as $S = e^{-Z}$, and the annual mortality rate (A) is $A = 1 - S$.

We evaluated the diet of each trout species collected from DCR by graphing the percentage of fish, by 10-mm size category, which were found to have GS in their stomach samples. We combined the data collected from all years (2014 - 2018) to increase sample size.

Angler exploitation rates and associated CIs (based on reported tags) were calculated in 2018 for TT and RBT using the methodology described in Meyer et al. (2010). Angler exploitation (percent of stocked fish harvested) and total use (percent of stocked fish caught) of TT and RBT were compared between fishes stocked in 2018, 2016 - 2017, and 2011 - 2012. To facilitate comparisons, 90% CI were calculated around exploitation and total use rates 365 and 730 days post-stocking. To show how rapidly the tagged/stocked TT and RBT were caught, this data was displayed graphically with percent of reported tags on the y-axis and days-at-large on the x-axis. To evaluate which size classes of stocked TT and RBT were being caught by anglers, in a length frequency graph of tagged TT and RBT, we showed what portion of each 10 mm length bin that was reported as being caught. We also compared mean TL of all TT tagged in 2016 - 2018 and RBT tagged in 2011 - 2018 to the mean TL (at time of tagging) of tagged fish reported as caught by anglers using two-tailed *t*-tests (assuming equal variance) with a significance level of $\alpha = 0.10$.

RESULTS

GOLDEN SHINER

Mean GS CPUE (± 90% CIs) in 2018 was 92 fish/transect (± 12) which was higher than was observed in 2014 and 2017 (Figure 5). However, ANOVA indicated that the CPUE did not significantly differ among years (F = 1.44; df = 2, 57; P = 0.245).

Golden Shiner mean TL (± 90% CIs) was 81 mm (± 0.04). Significant in GS mean TL occurred between years differences (F = 255.26; df = 2, 3,856; P < 0.01). Tukey-Kramer tests showed that mean TL did not significantly differ between 2017 and 2018, but both 2017 and 2018 were significantly larger than 2014 (Table 1; Figure 6).

TIGER TROUT

Approximately 2,500 catchable-size TT have been stocked into DCR annually since 2016 (Figure 9). This equates to a stocking rate of 83 fish/ha. The stocking rate for catchable trout (TT and RBT) in DCR is 304 fish/ha lower than the regional mean (520 fish/ha) for Clearwater Region's lowland reservoirs (Table 2).

In 2018, TT were collected through three surveys, including summer and fall electrofishing, and fall gill-netting. Catch, CPUE, and mean TL for each technique are summarized in Table 3. Catch rate of TT was the highest for fall electrofishing, but mean TL was highest for fall gill-netting.

The mean electrofishing CPUE for TT during the fall survey was 162 fish/h (± 47), the highest observed in all three years (Table 3 and Figure 10). Based on ANOVA testing, significant differences in TT CPUE occurred among years (F = 6.66; df = 2, 15; P < 0.01). In 2018, mean TT CPUE was significantly different than 2017 (P = 0.006), but not 2016 (P = 0.254; Table 4). The mean TL of TT during the fall electrofishing survey was 291 mm (± 1; Figure 11), and did not significantly differ among years (F = 0.35; df = 2, 355; P = 0.245; Table 4).

Relative weights of TT of all sizes sampled during fall electrofishing ranged from 61 to 101, with a mean of 79, while TT > 350 mm had a mean relative weight of 78 (Figure 12). Mean relative weight of TT during fall electrofishing surveys (2016 - 2018) did not significantly differ among years for TT of all sizes (F = 0.01; df = 2, 396; P = 0.124) or for TT > 350 mm (F = 0.49; df = 2, 28; P = 0.493; Table 5). Mean relative weight of TT > 350 mm (W_r = 82) was higher and significantly different than TT < 350 mm (76; P = 0.056).

Otoliths were collected from 29 TT sampled by summer electrofishing (Table 3). Of these fish, 26 were determined to have been stocked at catchable-size, while three were determined to have been stocked as fingerlings. Tiger trout stocked at catchable-size averaged 240 mm at age-1, and 286 mm at age-2 (Table 6). Annual growth for TT stocked at catchable-size was 45 mm from age-1 to age-2. The mean TL of age-1 and age-2 catchable-size TT at the time of stocking (age-1) were smaller than the average TL of TT stocked in 2017 (271 mm) and 2016 (286 mm). Tiger trout stocked at fingerling size averaged 151 mm at age-1, 195 mm at age-2, and 246 mm at age-3.

For TT stocked at catchable-size, annual instantaneous mortality (Z) was -0.7732 for fish age-2 to 3 (R^2 = 1.0). Thus, the estimated annual survival rate (S) was 46%, and total annual mortality (A) was 54%. Mortality was not estimated for TT stocked as fingerlings due to only collecting one age group.

We conducted diet analysis of 123 TT collected from DCR from 2014 to 2018. Zooplankton were the most commonly identified item, and were present in 68% of TT (Figure 7). Golden Shiner were present in 28% of TT. No GS were present in TT < 250 mm (Figure 8).

In 2018, angler exploitation and total use of stocked TT through 365 days after release were 11.9% and 15.4%, respectively. No additional tags were reported in the second year following stocking. For TT stocked in 2018, exploitation and total use were lower than in 2016 or 2017, although the 90% CIs overlapped in all cases (Table 7). Thirteen of the 200 TT tagged in 2018 were reported as caught by anglers, with 100% of tags reported within 150 days of stocking (Figure 13). In contrast, 74% of tagged TT caught from the 2017 tagging, and 29% from the 2016 tagging were caught within 150 days of stocking (Figure 13).

Stocked TT that were tagged from 2016 to 2018 had a mean TL of 271 mm (± 3) at the time of tagging (Figure 14). Tiger trout reported caught by anglers were larger than average at the time of stocking. Tagged TT reported as caught by anglers had a mean TL of 285 mm (± 9) at the time of tagging, which was significantly larger ($P < 0.001$). About 30% of the TT tagged from 2016 to 2018 were < 250 mm, whereas 12% of the TT reported as caught were < 250 mm at the time of tagging.

RAINBOW TROUT

Approximately 8,200 catchable-size RBT are stocked into DCR annually since 2015 (Figure 9). This equates to a stocking rate of 273/ha (Table 2). In 2015, stocking rates of RBT were reduced by about 61% (compared to the mean of 2004 - 2014) when we switched to stocking magnum-size (350 mm average) RBT and stocking of catchable-size TT was initiated.

In 2018, RBT were collected through fall electrofishing and fall gill-netting. Catch, CPUE, and mean TL by sampling method are summarized in Table 3. Catch of RBT by electrofishing was over two times higher than with gill nets, while mean TL was greater for gill nets.

The mean CPUE for RBT sampled during the fall electrofishing survey was 49 fish/h (± 16), the lowest for all surveys (Figure 10). Based on ANOVA testing, significant differences ($F = 5.55$; $df = 2, 15$; $P < 0.001$) in RBT CPUE occurred among years (Table 4). Mean CPUE in 2018 was lower and significantly different than 2017, but not significantly different than 2016. The mean TL of RBT sampled during the fall electrofishing survey was 312 mm (± 1; Figure 15). Based on ANOVA testing, significant differences ($F = 7.85$; $df = 2, 290$; $P < 0.001$) in RBT mean TL occurred among years (Table 4). In 2018, mean TL was longer and significantly different than 2017, but not significantly different than 2016 (Table 4).

We conducted diet analysis on 214 RBT. Zooplankton were the most commonly identified item, and were present in 84% of RBT (Figure 7). Golden Shiner were present in 7% of RBT. No GS were present in RBT < 260 mm (Figure 8).

Angler exploitation (33.1%) and total use (41.6%) of stocked RBT through 365 day-at-large was higher in 2018 than 2011 or 2012, although the 90% CIs overlapped in some cases (Table 7). There was no additional exploitation or total use beyond 365 days-at-large. Thirty-five of the 200 RBT tagged in 2018 were reported as caught within 300 days of being stocked. Ninety-one percent of these fish were caught within 100 days of stocking (Figure 16). In contrast, 55% of tagged RBT caught from the 2011 tagging, and 71% from the 2012 tagging were caught within 100 days (Figure 16).

Stocked RBT tagged in 2018 had a mean TL of 302 mm (± 4) at the time of tagging (Figure 17). Tagged RBT reported as caught by anglers had a mean TL of 307 mm (± 9) at the time of tagging, which was not significantly different (P = 0.204).

BROOK TROUT

Approximately 2,500 fingerling size BKT are stocked into DCR annually (Figure 9). This equates to a stocking rate of 83/ha.

In 2018, BKT were collected through fall electrofishing and fall gill-netting. Catch, CPUE, and mean TL for each technique are summarized in Table 3. Catch of BKT by fall gill-netting was two times higher than with fall electrofishing, while mean TL was also larger for gill nets.

The mean electrofishing CPUE for BKT was 3 fish/h (± 3), the lowest for all surveys (Figure 10). Based on ANOVA testing, significant differences (F = 4.02; df = 2, 15; P = 0.040) in BKT CPUE occurred among years (Table 4). Mean CPUE in 2018 was lower and significantly different than 2016, but not significantly different than 2017 (Table 4). In 2018, mean TL of BKT sampled during the fall electrofishing survey was 275 mm (± 53). ANOVA indicated that this average TL was not statistically different than what was sampled during fall electrofishing surveys conducted in 2016 or 2017 (Table 4).

Diet analysis was conducted on 44 BKT. Zooplankton were the most commonly identified item, and were present in 73% of BKT (Figure 7). Golden Shiner were present in 16% of BKT. No GS were present in BKT < 240 mm (Figure 8).

WESTSLOPE CUTTHROAT TROUT

In 2018, no WCT were collected.

DISCUSSION

GOLDEN SHINER

From 2014 to 2018, GS abundance has remained stable, while mean TL has increased. Stocking predators is often used to reduce the abundance of prey species (Koenig et al. 2015; Winters and Budy 2015; Smith et al. 2021). For DCR, the consistent catch rates suggests that DCR has reached equilibrium with impacts of TT predation on the GS population. The increase in GS mean TL may be due to predation. Predators are known to directly impact prey size structure, often resulting in an increase in average prey length when smaller prey is targeted (Tonn et al. 1992; Persson et al. 1996; Nilsson and Bronmark 2000). This has been shown to occur with GS after the introduction of predators such as Lake Trout *Salvelinus namaycush*, Largemouth Bass *Micropterus salmoides*, and Northern Pike *Esox lucius* (He and Kitchell 1990; Mittelbach et al. 1995; Johannes et al. 2011). Tiger trout have been found to become piscivorous at sizes > 270 mm (Miller 2010; Hand et al. 2020). The TT stocked into DCR from 2016 to 2018 averaged ~270 mm suggesting half of these fish were of the size where they could immediately prey upon GS. Previous diet analysis of TT in DCR indicated that GS consumed were mostly < 100 mm, though the GS were all in poor condition and difficult to measure accurately (Hand et al. 2020). In eastern Washington lakes, studies indicates that TT prey averaged 64 mm (Miller 2010). In contrast, mean TL was 139 mm for Utah Chub *Gila atraria* consumed by TT in Scoville

Reservoir, Utah, (Winters 2014). While this indicates TT are capable of consuming larger prey than they are in DCR, the TT in Scoville Reservoir were larger (mostly > 350 mm). However, the shift in the GS population length frequency toward larger fish may not be exclusively a result of predation. In 2014, the GS population in DCR was relatively new and therefore skewed towards younger fish as the population became established through natural reproduction. In 2017, and 2018, we would expect mean TL to be longer due to multiple years of growth and reproduction. This would explain the increase in mean TL and higher abundance of larger fish. Our data suggests the observed decline in abundance of GS < 80 mm and increased average size is likely due to a combination of a more mature population and predation by TT. It also suggests TT of the sizes present in DCR during 2018 may not be as effective at consuming large GS, as they are likely faster and more evasive, or that there are not enough larger TT to impact the abundance of larger GS. Overall, it appears that TT are not suppressing GS abundance or size enough to warrant reduced stocking rates based on this factor. At this time, we recommend surveying the GS population again in 2019. If CPUE and mean TL remain consistent, we can discontinue annual surveys for GS.

TIGER TROUT

Tiger trout abundance and their size structure observed in 2018 sampling efforts has not changed significantly from 2016 when catchable fish were first stocked. The following paragraphs discussing diet, relative weights, age, growth, mortality, and exploitation will explore and discuss why we are not seeing improvements in abundance and size structure of these fish despite consistent stocking over three years.

Diet Analysis

Diet analysis indicated that TT are successfully preying upon GS, although no GS were found in trout of any species < 240 mm in TL. Therefore, TT don't appear to begin preying upon GS until they reach this size. This minimum length of piscivory was lower than the 280 - 300 mm minimum observed for TT in other studies (Miller 2010; Winters 2014). However, approximately 18% of the TT stocked annually are < 240 mm in TL. These fish will likely need a year in the reservoir before they are large enough to begin preying on GS. The presence of GS in almost 30% of TT stomachs indicates they are effectively utilizing GS as a prey source, even though our analysis suggests they may not be affecting GS abundance.

Relative weight

There was no significant change in mean relative weight of TT for fall electrofishing surveys conducted from 2016 to 2018 for either all sizes of TT, or only those individuals > 350 mm. In general, fall relative weights of TT in DCR 78) were lower than those observed in other studies (84 - 94; Miller 2010; Winters 2014). With no change in TT relative weight over time, our data indicates that DCR is not being overstocked, as we would expect relative weights to decline if there were too many fish competing for food resources.

Mean relative weight of TT > 350 mm was higher and significantly different than TT < 350 mm for each year of our survey (2016 - 2018). An evaluation of TT relative weight from reservoirs in eastern Washington were mixed, with TT > 350 mm having higher relatives weight in some reservoirs, but lower relative weight in others (Miller 2010). In Wallace Lake, Idaho, and Scofield Reservoir, Utah, TT > 350 mm had similar relative weights to smaller fish (Winters 2014; Messner and Schoby 2019). The higher relative weights for larger size TT in DCR indicates that they are feeding and improving their body condition at larger sizes, and can potentially provide the put-grow-take fishery that was the intent of this stocking program. However, the overall low relative weight values and slow growth suggest food limitations or overstocking. With low TT relative weights in our surveys and other studies (Winters 2014; Messner et al. 2017; Messner and Schoby 2019), we should also consider the possibility that these relative weights are actually normal for TT. The use of the length-weight equation for Brown Trout may not be appropriate. If we see similar relative weights in future surveys with higher growth rates, we would recommend exploring the development of a TT-specific length-weight equation that would be more appropriate for Idaho reservoirs. Alternately, if further analysis confirms the low growth rates, we may need to consider reduced stocking rates.

Age, growth, and mortality

Annual survival of TT in DCR was estimated at 46%. We could not find annual mortality rates for stocked TT in the literature, but the 54% estimated for TT in DCR was lower than the mean annual mortality rate of 84% estimated for hatchery trout in 12 Idaho reservoirs (Meyer and Schill 2014). We caution that our mortality estimate was based on only two year classes and limited data. Due to the limited sample size of TT aged in 2018, we recommend collecting additional aging structures in 2019 to better assess growth, mortality, and carryover. Additionally, we recommend utilizing gill nets in the fall to collect a more robust sample. This will allow us to evaluate whether the slow growth rates seen in 2018 were due to our small sample size.

Annual growth of age-1 to age-3 TT ranged from 44 to 51 mm. This is similar to growth observed in Wallace Lake, Idaho, and Black Lake, Washington, but lower than the 74 mm observed in Fish Lake, Washington (Miller 2010; Messner and Schoby 2019). Mean back-calculated TL of TT stocked as catchable at ages 1 - 2 were similar or higher than estimated for eastern Washington lakes (Miller 2010). However, it must be noted that TT were stocked as fingerlings in eastern Washington lakes, as opposed to catchable-size in DCR. In contrast, the mean back-calculated TL of TT at ages 1 - 3 for the three fingerlings collected in DCR were similar or lower than estimated for eastern Washington lakes (Miller 2010). This data indicates that although some TT stocked as fingerlings survived to catchable-size, stocking TT at catchable-size is more successful and beneficial for this fishery.

The mean TL at age for TT collected for age and growth analysis was smaller than the mean size of fish stocked into DCR. This phenomenon is commonly seen in fisheries, as angler prefer to catch and harvest the larger individuals in a population (Ricker 1992; Quist et al. 2012). Therefore, it appears that anglers are harvesting the larger TT in DCR each year. However, our data also shows that the smaller remaining fish are surviving and growing to sizes that can utilize GS as a food source if they carryover, and may then reach harvestable size.

Exploitation

The angler total use rate for TT through 365 days has not increased significantly for fish stocked from 2016 to 2018. The estimates (9.8 to 15.4%) were within the range seen at Wallace Lake, Idaho (10.7 - 20.5%) from 2015 to 2017 (Messner and Schoby 2019), and were similar to RBT stocked in DCR in 2011 and 2012 (~13.5%). However, these total use rates would be considered low, as they are below the Idaho statewide average in 2011 - 2012 (~28%; Cassinelli 2014), and the mean for all Clearwater Region reservoirs in 2012 (~24%) for all trout species (Hand et al. 2016). We expected angler effort and exploitation of TT to increase as this fishery became better known. Tiger trout being more difficult to catch than we hoped could explain for the lower angler total use rates. The total use rate for RBT stocked in DCR in 2018 (33%) suggests TT are more difficult for anglers to catch than RBT. However, the TT stocked into DCR were smaller than the stocked RBT. Higher catchability of larger fish has been well documented with stocked trout (Wiley et al. 1993; Cassinelli et al. 2016).

The timing of when tagged TT were reported as being caught has changed considerably from 2016 to 2018. Tag return studies for stocked catchable-size RBT in Idaho indicate that 75% occur within 122 days of stocking (Cassinelli and Meyer 2018), while in Wyoming "most" (as described by Wiley et al. 1993) are caught within two months of stocking. This can primarily be attributed to stocking when angling effort is high (spring/early summer) and fishing conditions are good (Wiley et al. 1993; Walters et al. 1997; Cassinelli and Meyer 2018). However, in DCR, TT tag returns have occurred more quickly each year, and in 2018 no tags were returned beyond 161 days-at-large.

Summary

Tiger Trout were found to maintain their body condition after being stocked from one year to the next, suggesting that food sources are available to maintain body condition, and stocking rates are not excessive. However, growth rates are considered to be low (~ 50 mm/year) compared to the > 70 mm mean annual growth observed for catchable-size RBT in Idaho reservoirs (Koenig and Meyer 2011). The combination of low growth rates and higher annual mortality rates (54%) suggests that the maximum potential for TT in DCR is about 500 mm. Because few TT are caught after the initial year of stocking, this fishery has become more of a put-and-take fishery. Despite that, stocking catchable-sized TT in DCR has created a unique fishing opportunity that anglers are utilizing and targeting. Due to annual fluctuations in CPUE, and limited data on annual survival, we recommend conducting an additional survey in fall 2019 to compare with previous surveys and finalize our evaluation. Catch of TT was 2.5 times higher from electrofishing than gill nets. While not directly comparable, this indicates that electrofishing is an effective method for the collection of TT, eliminating a potential source of sampling bias had TT been found to be primarily pelagic at the time of fall sampling. Additionally, due to the continued decline in days-at-large for TT, and limited understanding of TT growth and mortality, additional exploitation as well as age, growth, and mortality data are needed to assess whether a put-grow-take TT fishery in DCR is a reasonable expectation.

RAINBOW TROUT

The mean CPUE for RBT during the fall electrofishing survey was lower and significantly different than 2017, whereas stocking densities remained consistent between years. While this may be just annual variation in catch, it is likely at least partially due to the high angler exploitation rate in 2018. Additionally, with < 10% of tag returns occurring beyond 100 days at large suggests that a low percentage of RBT stocked survived until the fall electrofishing survey.

The mean TL of RBT sampled between years (2016 - 2018) significantly differed. The mean size of hatchery RBT can differ by 50 mm due to annual differences in growth, date of stocking, and hatchery of origin (Chris Jeszke, *personal communication*; IFWIS 2021a). However, the average TL at stocking was the same in 2017 and 2018, and only 5 mm longer in 2016, indicating this was not a factor in why significant differences in mean TL were observed between years (IFWIS 2021b). The RBT fishery in DCR is put-and-take, and based on previous surveys carryover averages approximately 6% from year to year (Hand et al. 2016). Thus, the difference in mean TL was likely related to annual variation in growth between years as opposed to sampling larger fish that have been in the reservoir multiple years. With stocking rates below the regional average, there is little risk of overstocking the reservoir if carryover remains low. It is worth noting that the mean lengths of RBT in our fall surveys is similar to the mean length of fish stocked in the spring. This may indicate some level of food limitation or overstocking. However, it is could also be a result of anglers' preferences in keeping larger fish, which would reduce the average size of fish remaining in the reservoir (Aday and Graeb 2012). We recommend evaluating the zooplankton community to determine if GS are still reducing densities and size of Daphnia.

The mean TL of RBT sampled in fall electrofishing was 12 mm larger than the mean TL of RBT stocked in the spring. Studies of catchable-size RBT stocked in Idaho indicate growth of 70+ mm from spring stocking to fall surveys (Koenig and Meyer 2011). Thus, the RBT stocked into DCR are growing far less than would be expected, suggesting a lack of food resources and competition. With zooplankton found in 84% of RBT stomachs, it does not appear that there is a lack of food quantity, although the zooplankton present could be of poor quality. We recommend evaluating the zooplankton population to determine if it is providing an adequate food source.

We do not believe there was a sampling bias from electrofishing as the size composition of RBT was similar to what was sampled by gill nets. Additionally, twice and many fish were collected through electrofishing as gillnetting in one night of sampling. Electrofishing therefore appears to be an effective method for the collection of RBT in DCR despite its poor capture efficiency in pelagic waters.

Diet analysis indicated that RBT are preying upon GS, although at a lower rate (8%) than TT (28%) or BKT (14%). With GS found in only 8% of RBT stomachs, it appears RBT are not utilizing GS as a primary food source. This is to be expected, as RBT primarily feed upon plankton and insects in lentic waters (Beauchamp 1990; Sigler and Zaroban 2018). Additionally, no GS were found in individuals < 260 mm in TL. This minimum length of piscivory was similar to that observed in Lake Washington (Beauchamp 1990), but smaller than the > 300 mm observed in some studies (Keeley and Grant 2001; Jensen et al. 2012). Therefore, RBT don't begin preying upon GS until they reach this size, and are not likely impacting the GS population.

The angler total use rate for RBT through 365 days in 2018 was 41.6%, which was significantly higher than in 2011 and 2012. This total use rate was higher than the mean for all Clearwater Region reservoirs in 2012 (~24%; Hand et al. 2016), and the statewide management goal of 40% (IDFG 2019). This increase in total use rate can be explained by the reduction in stocking densities and increased size of the fish that were stocked (Wiley et al. 1993; Cassinelli et al. 2016). With angler exploitation meeting our management goal we do not recommend any changes to the stocking strategy for RBT in DCR at this time. However, due to the significant increase in exploitation that occurred in 2018, we recommend one additional year of evaluation.

BROOK TROUT

Brook Trout mean electrofishing CPUE has declined significantly from 2016 to 2018. Predation by TT on BKT fingerlings may be part of this decline, as larger TT are known to utilize fish as a primary prey item (Miller 2010; Winters 2014). Additionally, fingerling BKT may also experience high mortality due to competition with GS for zooplankton. Low gill-net CPUEs in 2018 corroborate that these electrofishing estimates are real. Now that DCR has been stocked with catchable-sized TT for three years, predation on fingerling BKT is likely. Further evaluation of angler exploitation of BKT through a creel survey scheduled for 2019 will help assess if there is a fishery for BKT. If angler catch of BKT is low, we may want to consider eliminating this stocking from DCR.

Diet analysis indicated that BKT are preying upon GS, although at a lower rate (14%) than TT (28%). With GS found in only 14% of RBT stomachs, it appears BKT may not be utilizing GS as a primary food source. This is to be expected, as most trout species primarily feed upon plankton and insects in lentic waters (Winters and Budy 2015; Beauchamp 1990; Sigler and Zaroban 2018). Additionally, no GS were found in individuals < 260 mm in TL. This minimum length of piscivory was similar to that observed in Lake Washington (Beauchamp 1990), but smaller than the > 300 mm observed in some studies (Keeley and Grant 2001; Jensen et al. 2012). Therefore, BKT don't begin preying upon GS until they reach this size, and are not likely impacting the GS population.

<u>WESTSLOPE CUTTHROAT TROUT</u>

Since 2016, no WCT have been captured in surveys, suggesting they do not survive in DCR. Since they were last stocked in DCR in 2011 and have an expected life span of ~6-8 years (Sigler and Zaroban 2018), this was to be expected. We stopped stocking WCT fingerlings due to competition for food resources (primarily zooplankton) with GS. With three other trout species in DCR, all of which also compete for similar food resources, it is questionable whether renewed stocking would provide sufficient opportunity or increased angler satisfaction without detriment to the other species present. Thus, we do not recommend further stockings of WCT into DCR.

MANAGEMENT RECOMMENDATIONS

1. Continue to evaluate Golden Shiner CPUE and length distribution.

2. Collect otoliths (or other aging structure) from GS in future surveys to better understand their age, growth, and mortality, and how TT may influence these metrics.

3. Collect otoliths from TT sampled in the fall to better evaluate age, growth, mortality, and carryover. Sample TT in the fall as well to allow comparisons to previous years.

4. Evaluate zooplankton to determine if there have been changes in size and community structure post-introduction of catchable-size TT.

5. Evaluate angler exploitation of TT and RBT.

6. Stock TT at TL > 250 mm to provide a desirable fishery for anglers.

7. Evaluate potential mortality related to summer dissolved oxygen and temperature.

Table 1. Results from ANOVA comparing mean total lengths (mm; 90% CIs) among years (2014 - 2018) for Golden Shiner collected by electrofishing Deer Creek Reservoir, Idaho, and follow up pairwise comparisons using Tukey-Kramer tests. Significance was set at α = 0.10.

Year	Length	P-value
2014	69 (± 1)	
2017	81 (± 1)	< 0.001[1]
2018	81 (± 1)	

[1]Post-hoc Tukey-Kramer Test between years for mean length

Group 1	Group 2	P-value
2014	2017	< 0.001
2014	2018	< 0.001
2017	2018	0.986

Table 2. Stocking rates for catchable-size trout in Clearwater Region, Idaho, lowland lakes in 2018.

Reservoir	Size (ha)	Rate (fish/ha)
Deer Creek Reservoir	30	304
Elk Creek Reservoir	19	697
Mann Lake	25	328
Moose Creek Reservoir	14	1,271
Soldier's Meadow Reservoir	48	111
Spring Valley Reservoir	21	444
Waha Lake	33	136
Winchester Lake	44	871
Mean		520

Table 3. Summary of trout caught by different sampling techniques used in Deer Creek Reservoir, Idaho, in 2018. CPUE (90% CIs) is fish/h for electrofishing and fish/net for gill-nets. Length is mean total length in mm. Significance was set at α = 0.10.

	Fall electrofishing			Summer electrofishing			Fall gill nets		
Species	*n*	CPUE	Length	*n*	CPUE	Length	*n*	CPUE	Length
Tiger trout	162	162 (± 47)	291 (± 5)	29	29 (± 12)	294 (± 11)	69	17.3 (± 4)	359 (± 15)
Rainbow Trout	49	49 (± 16)	313 (± 7)	---	---	---	23	5.8 (± 2)	331 (± 1)
Brook Trout	3	3 (± 3)	276 (± 53)	---	---	---	6	1.5 (± 1)	331 (± 23)
Westslope Cutthroat Trout	0	0	---	---	---	---	0	0	---

Table 4. Summary of ANOVA comparing mean CPUE (fish/h; ± 90% CIs) and mean total length (mm) among years (2016 - 2018) for trout collected by fall electrofishing of Deer Creek Reservoir, Idaho. Where significance was detected follow up pairwise comparisons occurred using Tukey-Kramer tests. Significance was set at α = 0.10.

Species	Date	CPUE	P-value	Length	P-value
Tiger trout	2016	122 (± 24)	0.009[1]	292 (± 8)	0.704
	2017	74 (± 25)		293 (± 9)	
	2018	162 (± 47)		291 (± 5)	
Rainbow Trout	2016	100 (± 17)	0.016[2]	311 (± 5)	< 0.001[4]
	2017	144 (± 54)		299 (± 3)	
	2018	49 (± 16)		313 (± 7)	
Brook Trout	2016	16 (± 9)	0.040[3]	285 (± 20)	0.548
	2017	6 (± 4)		243 (± 49)	
	2018	3 (± 3)		276 (± 53)	
Westslope Cutthroat Trout	2016	2 (± 2)	n/a	266 (± 48)	n/a
	2017	0		---	
	2018	0		---	

[1]Post-hoc Tukey-Kramer Test among years for tiger trout CPUE

Group 1	Group 2	P-value
2016	2017	0.149
2016	2018	0.254
2017	2018	0.006

[2]Post-hoc Tukey-Kramer Test among years for Rainbow Trout CPUE

Group 1	Group 2	P-value
2016	2017	0.149
2016	2018	0.254
2017	2018	0.006

[3]Post-hoc Tukey-Kramer Test among years for Brook Trout CPUE

Group 1	Group 2	P-value
2016	2017	0.127
2016	2018	0.041
2017	2018	0.809

[1]Post-hoc Tukey-Kramer Test among years for Rainbow Trout mean length

Group 1	Group 2	P-value
2016	2017	0.002
2016	2018	0.921
2017	2018	0.007

Table 5. Summary of ANOVA comparing mean relative weight (W_r) of tiger trout (TT; all individuals and only those > 350 mm) sampled during fall electrofishing surveys in Deer Creek Reservoir, Idaho, from 2016 to 2018. Significance was set at α = 0.10.

Group	Year	W_r	*P* value
All TT	2016	79	0.124
	2017	75	
	2018	79	
TT > 350 mm	2016	83	0.493
	2017	84	
	2018	78	

Table 6. Back-calculated length at annuli of tiger trout collected by electrofishing Deer Creek Reservoir, Idaho, in 2018. Mean total length of tiger trout stocked in 2017 was 271 mm. Age group 3f are fish believed to have been stocked as fingerling. Mean length at age and annual growth only refer to age groups 1 and 2, fish which were stocked at catchable-size.

		Age		
Age group	*n*	1	2	3
1	14	243		
2	12	237	286	
3f	3	151	195	246
n	29	29	26	3
Length at age		240	285	
Annual growth			45	

Table 7. Angler exploitation (harvested fish) and total use (harvested and released fish) of tiger trout and Rainbow Trout stocked into Deer Creek Reservoir, Idaho, from 2011 to 2018, based on angler-reported, T-bar anchor tags. Estimates are reported for the first year (365 days at large) and cumulative over 2 years after stocking (730 days at large).

				Disposition						
Species	Tagging date	Tags released	Days at large	Harvested	Harvested b/c tagged	Released	Adjusted exploitation	90% CI	Adjusted total use	90% CI
Tiger Trout	June 2016	99	365	4	0	0	9.8%	7.4%	9.8%	7.4%
			730	7	0	0	14.6%	9.1%	17.1%	9.8%
	June 2017	200	365	11	0	1	13.1%	6.3%	14.3%	6.6%
			730	13	0	2	16.6%	6.9%	19.6%	7.1%
	June 2018	200	365	11	0	2	11.9%	6.0%	15.4%	6.9%
			730	11	0	2	11.9%	6.0%	15.4%	6.9%
Rainbow Trout	May 2011	397	365	21	0	0	12.6%	4.6%	12.6%	4.6%
			730	23	1	0	14.4%	5.2%	15.3%	5.4%
	May 2012	400	365	21	1	2	12.5%	4.6%	14.3%	4.9%
			730	32	1	2	22.2%	7.8%	24.0%	8.2%
	May 2018	200	365	28	2	5	33.3%	10.5%	41.6%	11.9%
			730	28	2	5	33.3%	10.5%	41.6%	11.9%

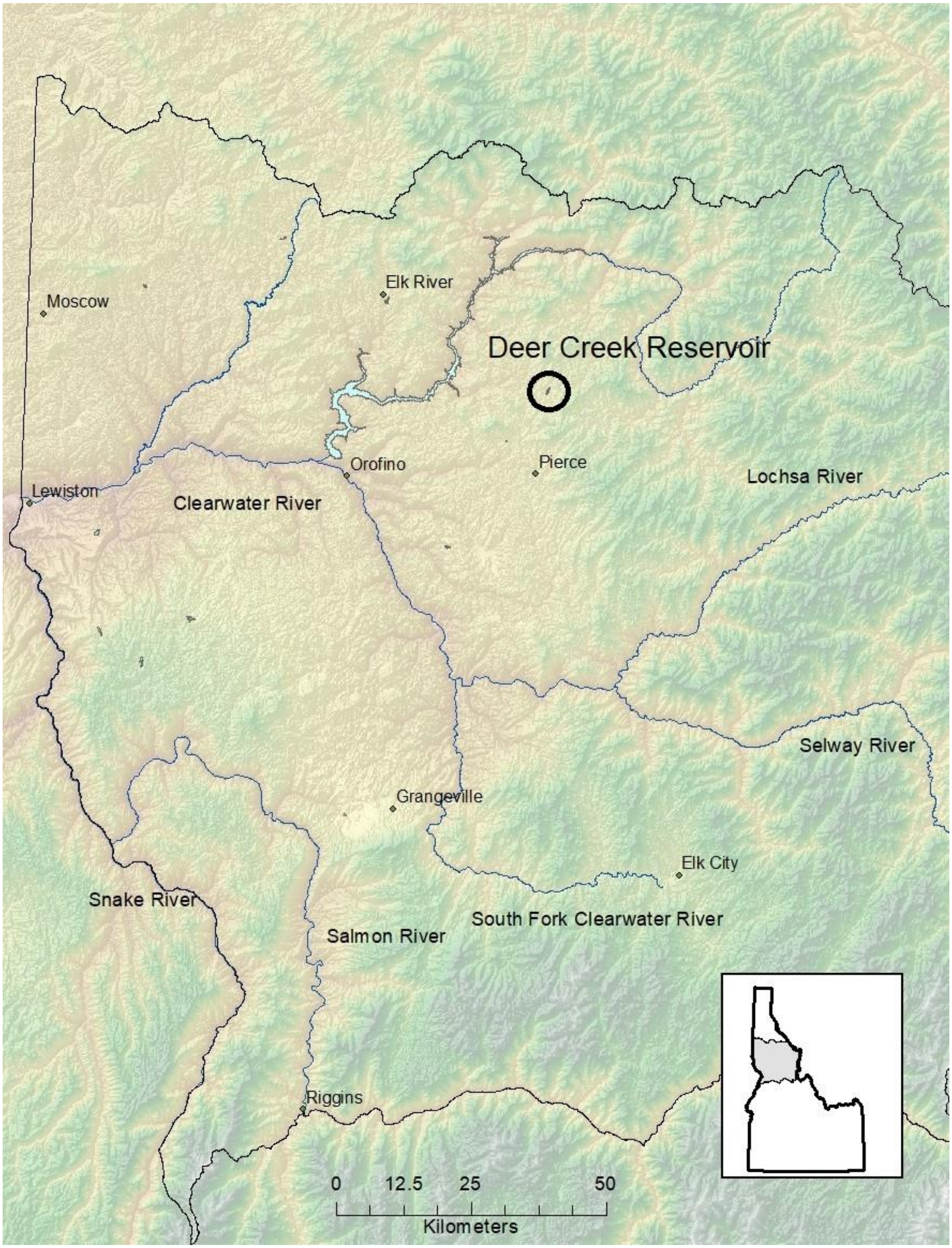

Figure 1. Map showing location of Deer Creek Reservoir, Idaho.

Figure 2. Locations of starting points for 50-m electrofishing transects used for Golden Shiner surveys in Deer Creek Reservoir, Idaho, from 2014 to 2018.

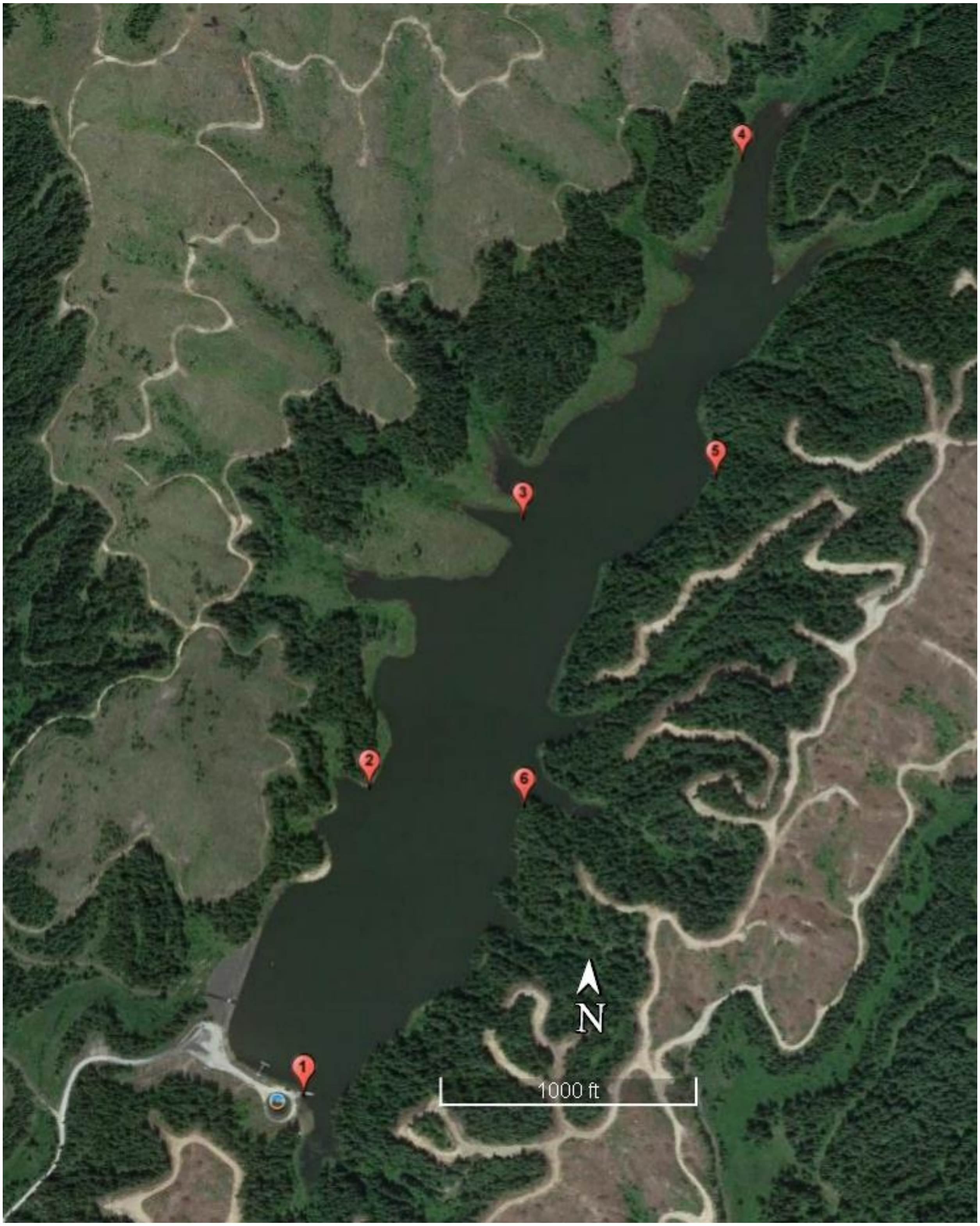

Figure 3. Locations of starting points for trout electrofishing transects on Deer Creek Reservoir, Idaho, for summer and fall surveys conducted from 2016 to 2018.

Figure 4. Location of four gill nets set in Deer Creek Reservoir, Idaho, on October 22, 2018.

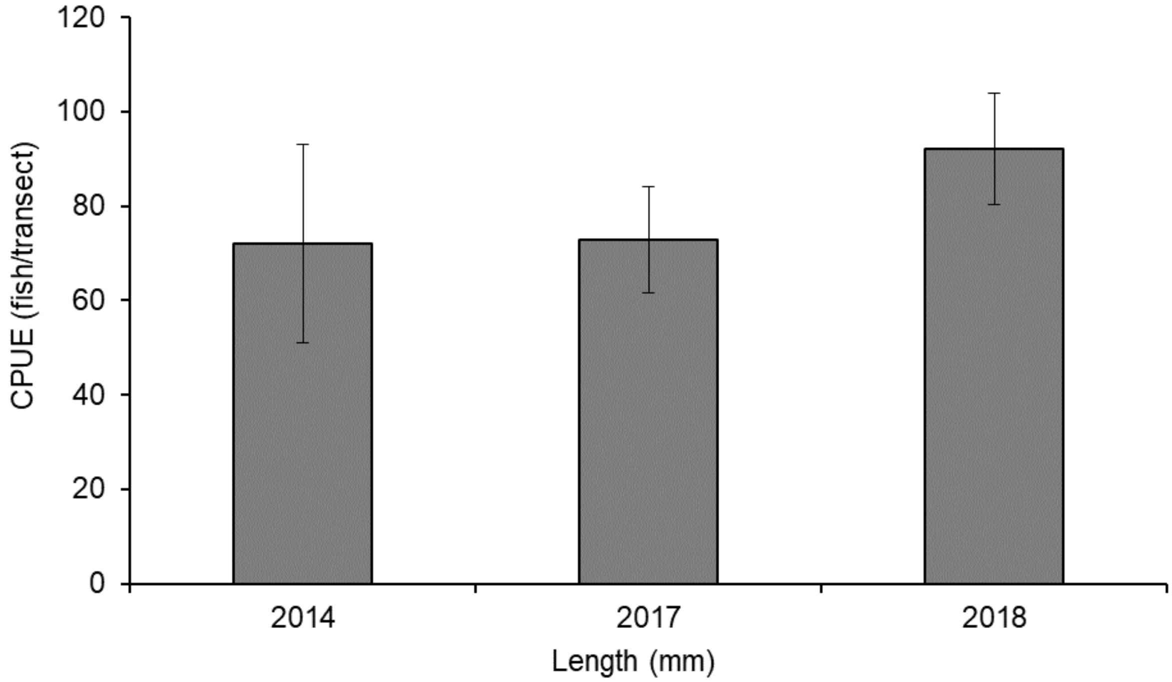

Figure 5. Comparisons of CPUE of Golden Shiner sampled by electrofishing Deer Creek Reservoir, Idaho, from 2014 to 2018. Error bars represent 90% CIs.

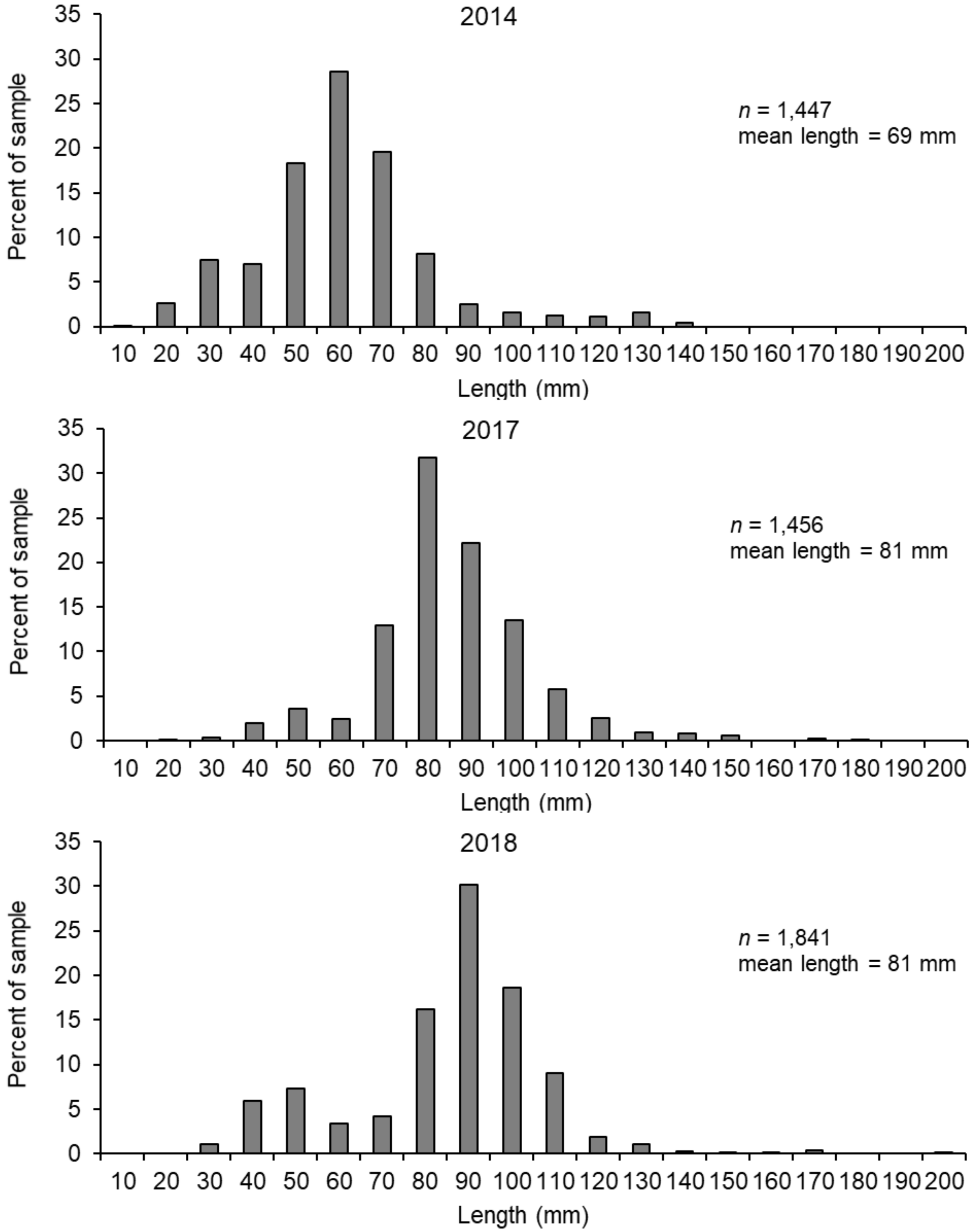

Figure 6. Relative length frequencies of Golden Shiner sampled by electrofishing Deer Creek Reservoir, Idaho, from 2014 to 2018.

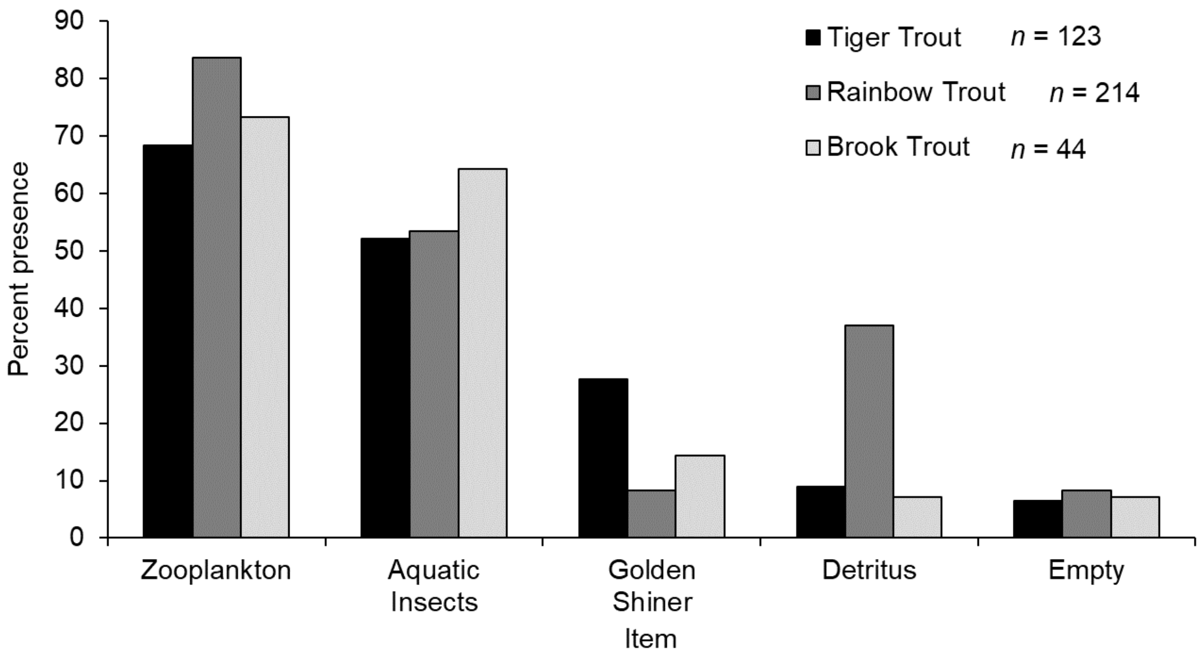

Figure 7. Contents of stomach samples from trout collected by gill nets in Deer Creek Reservoir, Idaho, from 2016 to 2018.

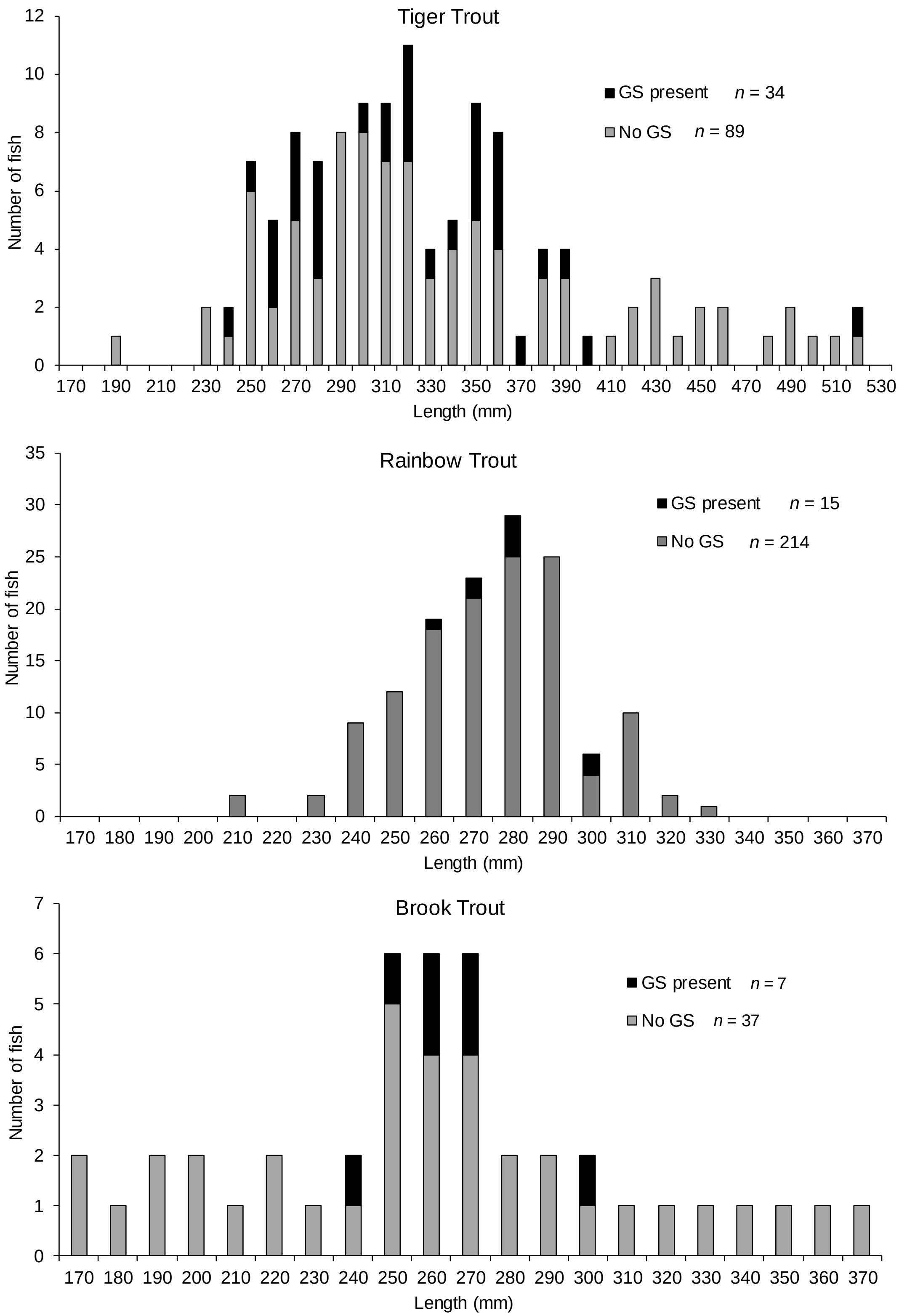

Figure 8. Length-frequency distributions of trout collected by gill nets in Deer Creek Reservoir, Idaho, from 2016 to 2018, with GS present in stomach samples versus those without GS present.

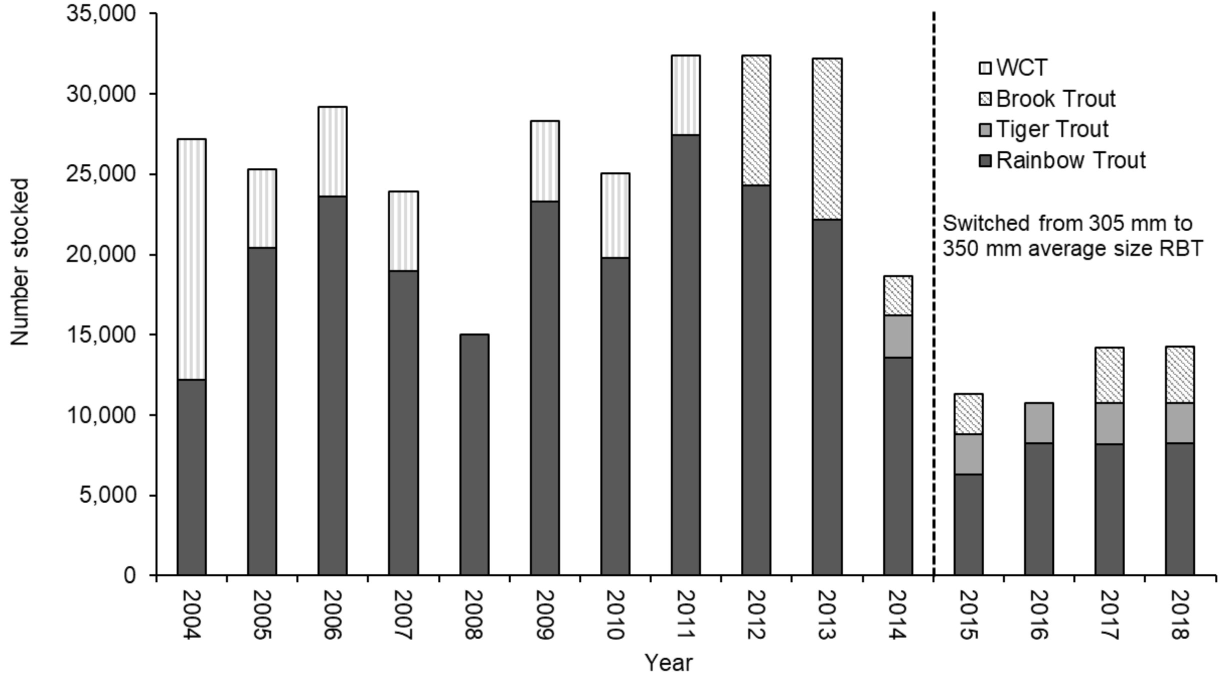

Figure 9. Number of trout (by species) stocked into Deer Creek Reservoir, Idaho, annually from 2004 to 2018.

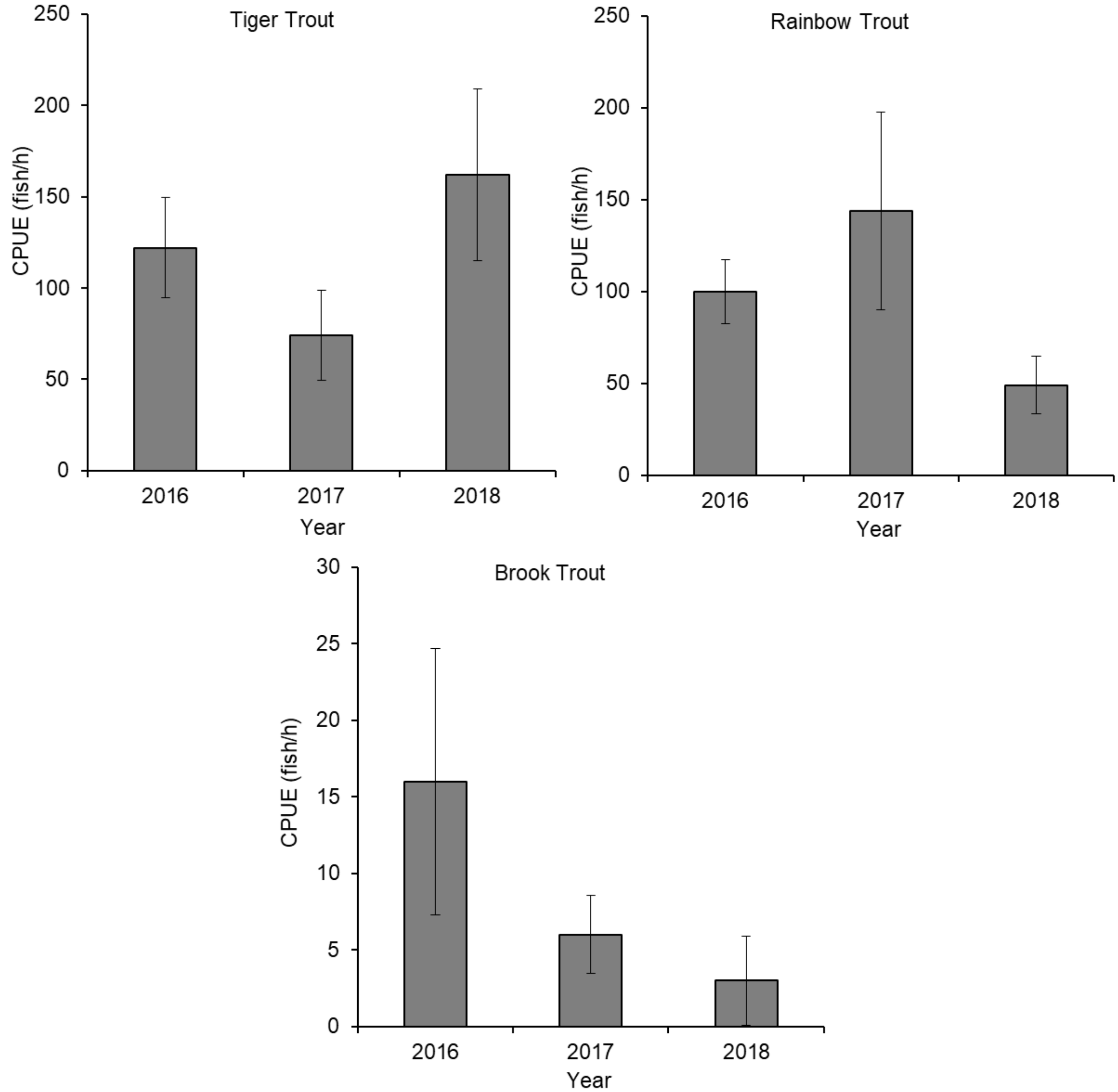

Figure 10. A comparison of CPUE among years (2016 - 2018) for tiger trout, Rainbow Trout, and Brook Trout sampled by fall electrofishing of Deer Creek Reservoir, Idaho. Error bars represent 90% CIs.

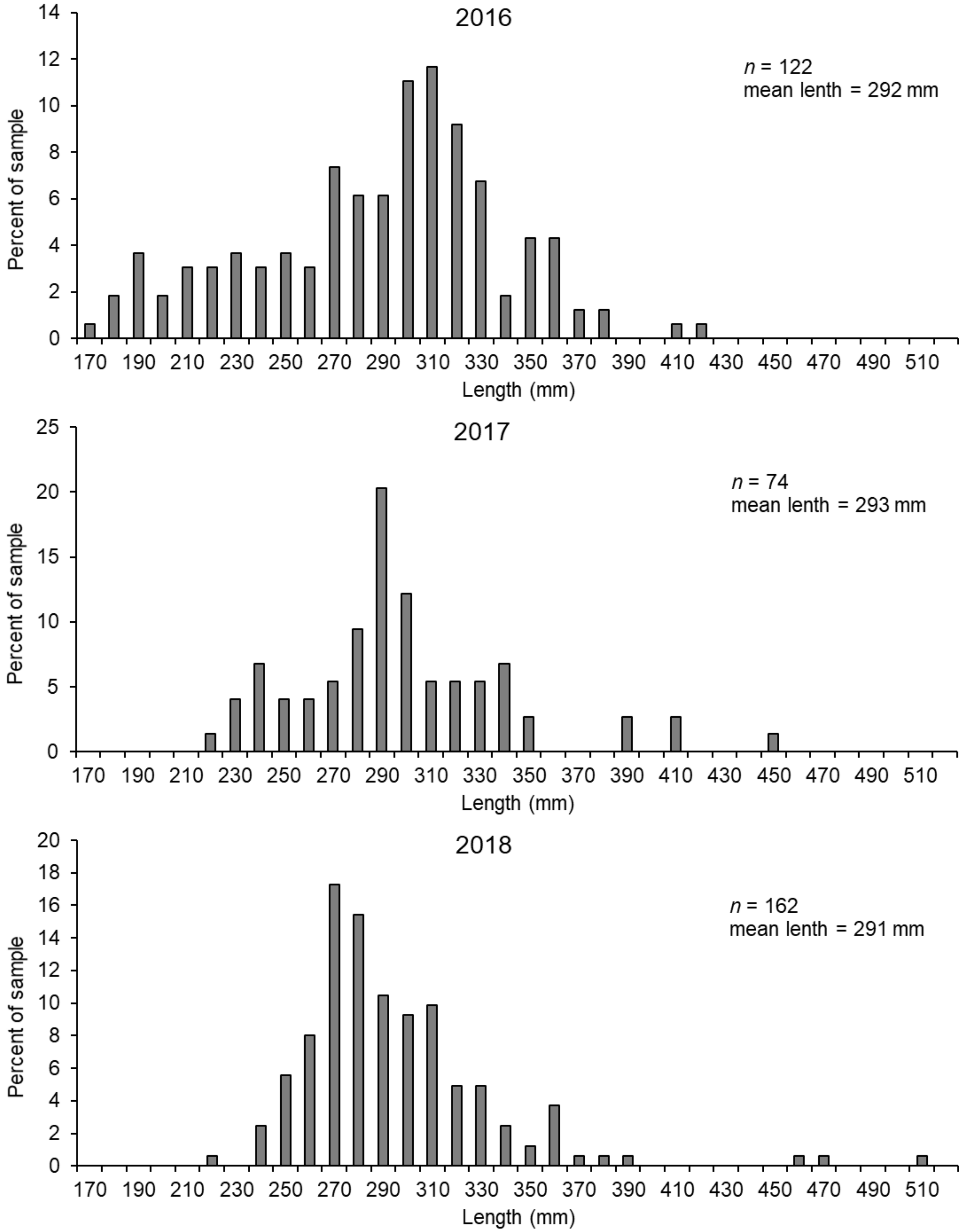

Figure 11. Relative length frequencies of tiger trout sampled by electrofishing Deer Creek Reservoir, Idaho, from 2016 to 2018.

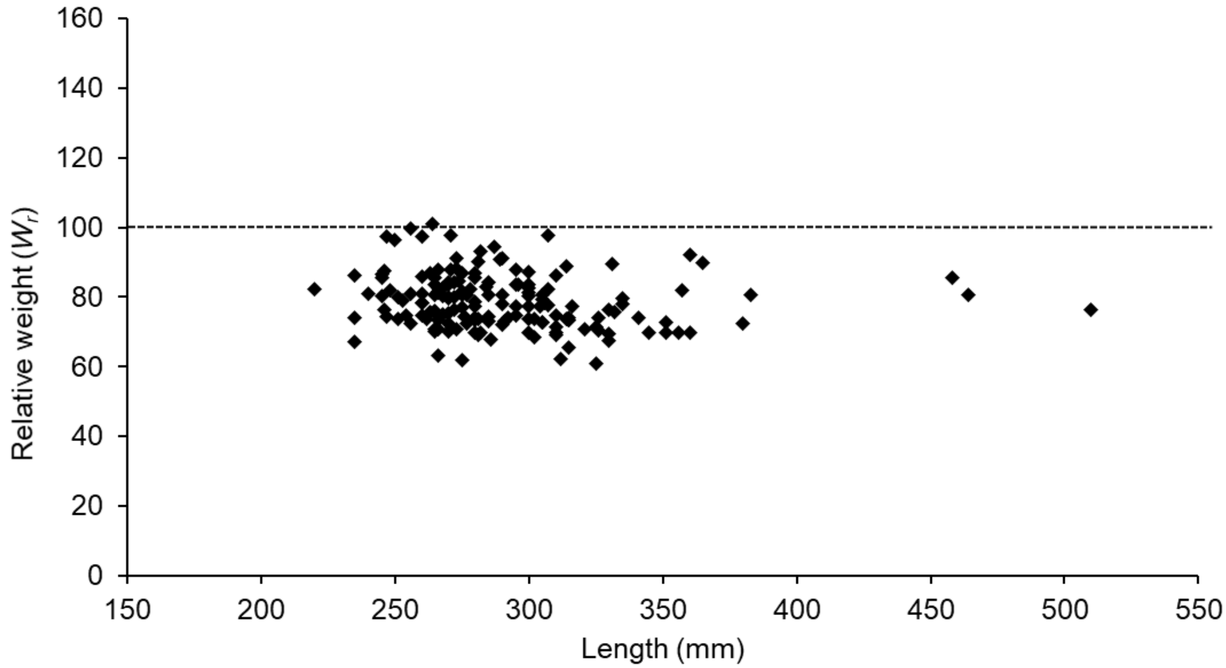

Figure 12. Relative weight of tiger trout sampled by fall electrofishing in Deer Creek Reservoir, Idaho, in 2018.

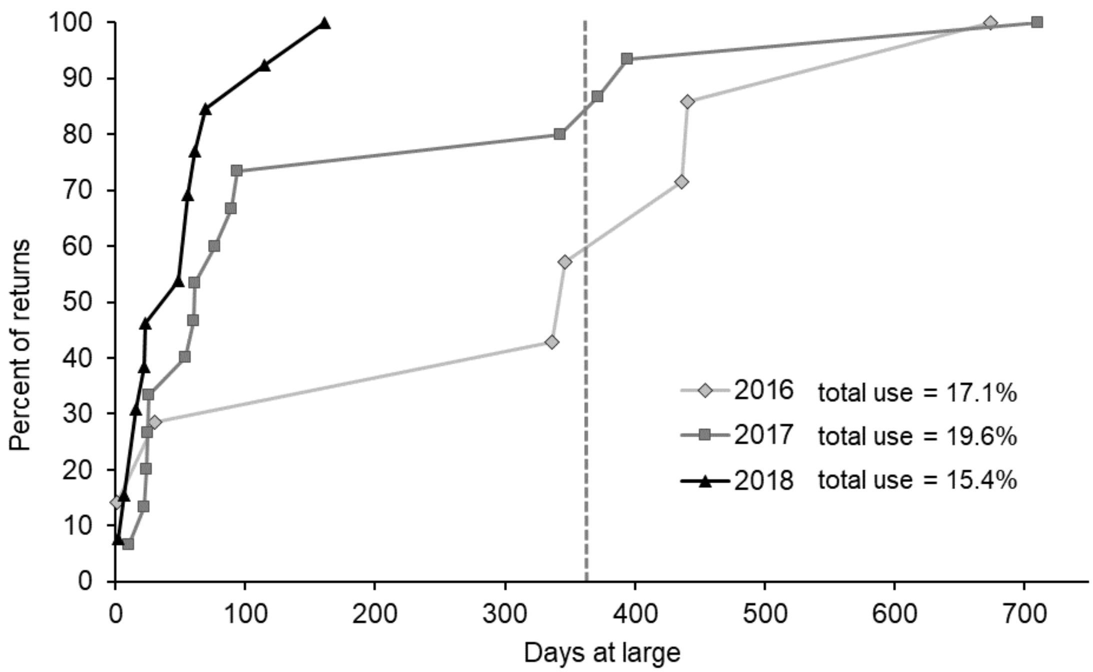

Figure 13. Days-at-large for tagged tiger trout reported as caught by anglers through the Tag You're It program, for fish tagged and stocked in 2016 (*n* = 7), 2017 (*n* = 14), and 2018 (*n* = 13) in Deer Creek Reservoir, Idaho, through 730 days-at-large. Vertical dashed line represents 365 days-at-large (1 year). Total use rate is through 730 days-at-large.

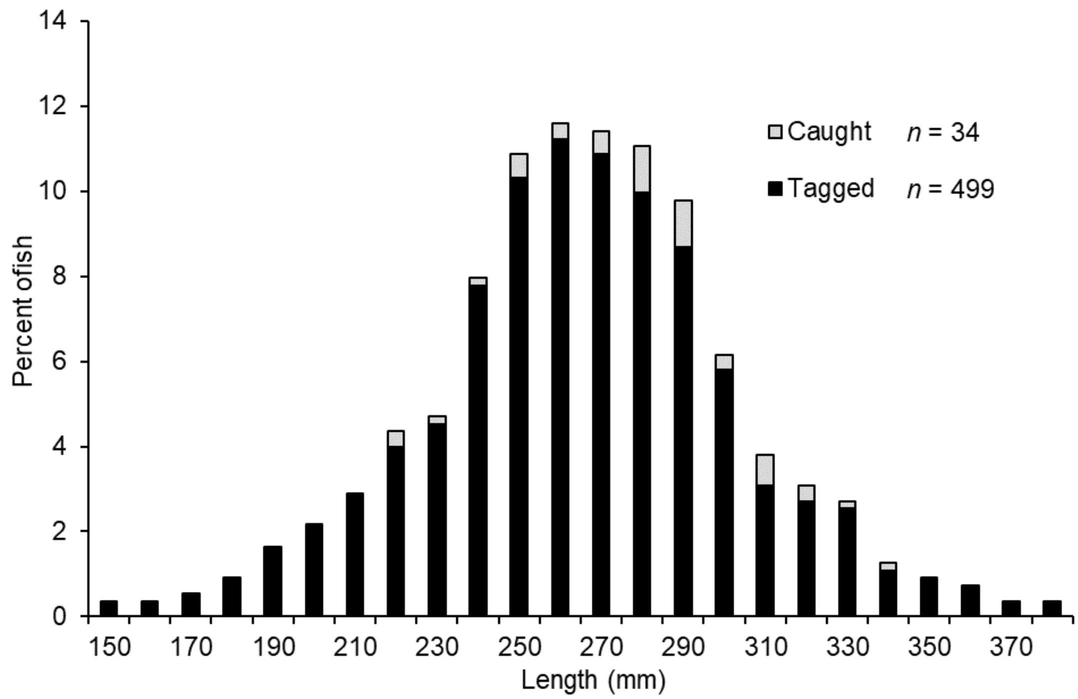

Figure 14. Relative length frequencies of tiger trout tagged and stocked in Deer Creek Reservoir, Idaho, from 2016 to 2018, with proportion of each size class caught by anglers (based on angler reported tags).

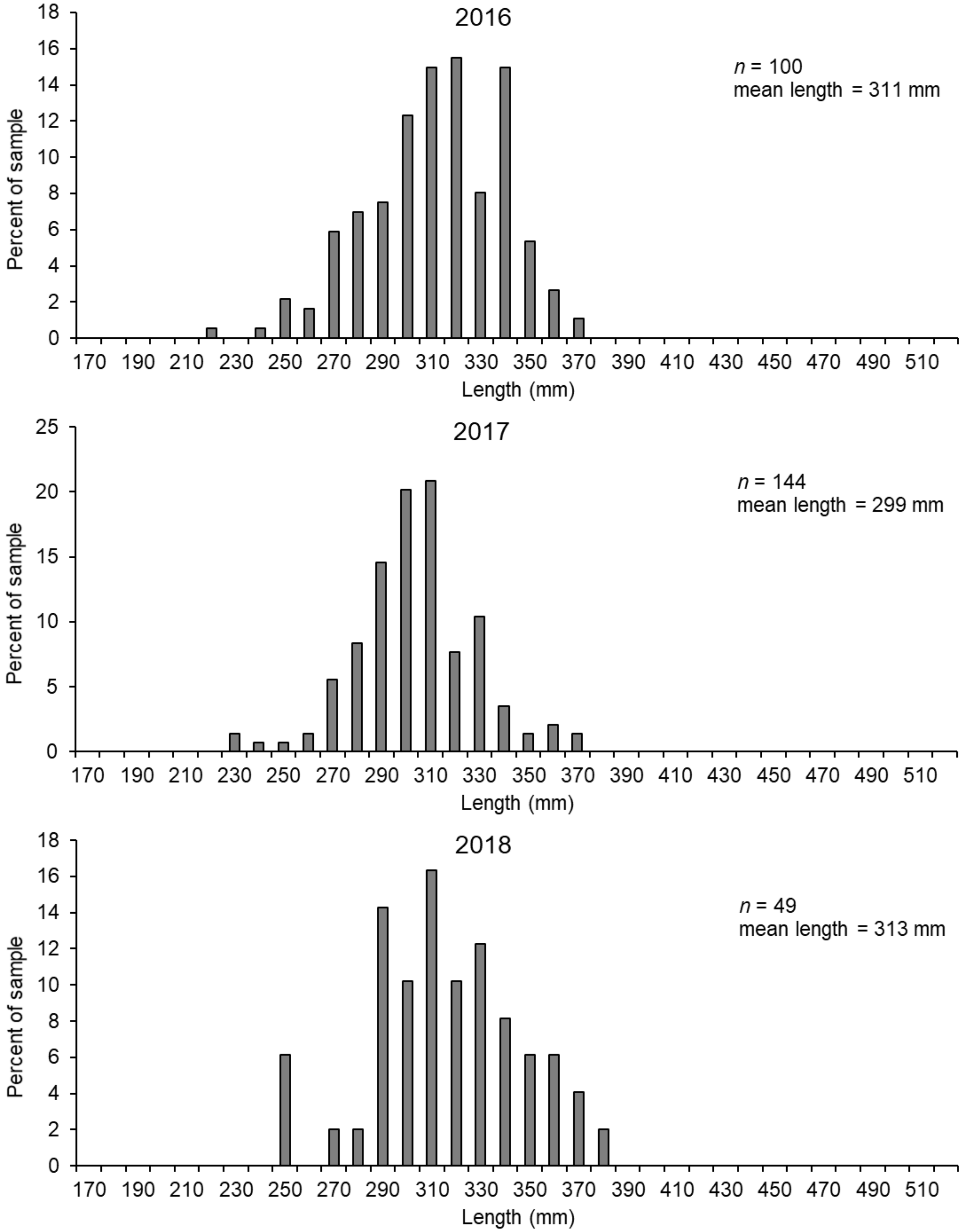

Figure 15. Relative length frequencies of Rainbow Trout sampled by fall electrofishing of Deer Creek Reservoir, Idaho, from 2016 to 2018.

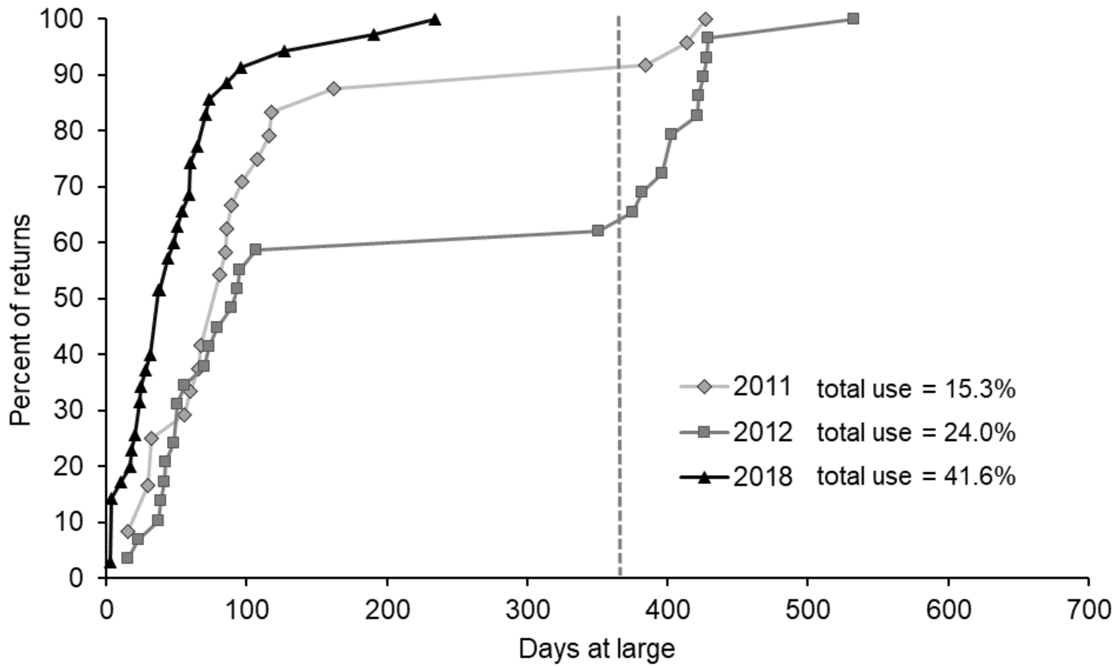

Figure 16. Days-at-large for tagged Rainbow Trout reported as caught by anglers through the Tag You're It program, for fish tagged and stocked in 2011 (n = 24), 2017 (n = 35), and 2018 (n = 35) in Deer Creek Reservoir, Idaho, through 730 days-at-large. Vertical dashed line represents 365 days-at-large (1 year). Total use rate is through 730 days-at-large.

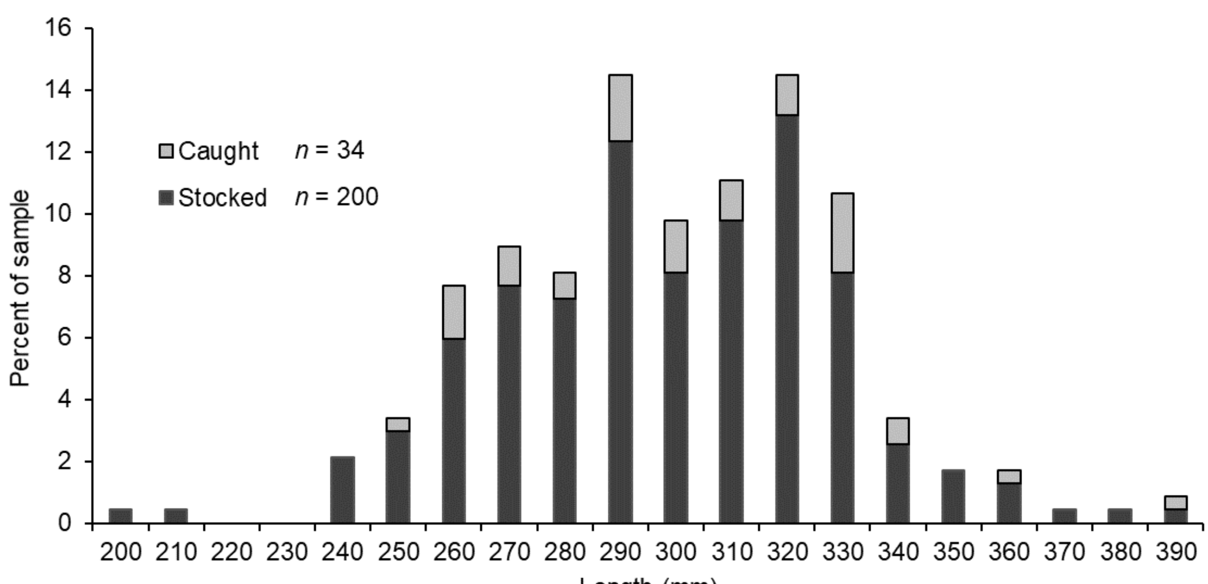

Figure 17. Relative length-frequency distribution of Rainbow Trout tagged and stocked in 2018 in Deer Creek Reservoir, Idaho, with proportion of each size class caught by anglers (based on angler reported tags).

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DWORSHAK RESERVOIR AND NORTH FORK CLEARWATER RIVER WILD TROUT INVESTIGATIONS

ABSTRACT

The North Fork Clearwater River (NFC) provides a popular fishery for Westslope Cutthroat Trout (WCT) *Oncorhynchus clarkii lewisi*, Rainbow Trout (RBT) *O. mykiss* and their hybrids (HYB). The NFC is also a core area for Bull Trout (BT) *Salvelinus confluentus,* currently listed as "threatened" under the Endangered Species Act (ESA). A component of all of these populations over winter in Dworshak Reservoir. Angling surveys were conducted near slack water on Dworshak Reservoir in early November of 2011, 2015, and 2018 to assess species composition, length, exploitation of WCT and hybridization between WCT and RBT. The catches were primarily WCT (73%), followed by BT (14%), HYB (8%), and RBT (5%). WCT catches increased after the first year concurrent with increased densities estimated from snorkel surveys, and BT catches declined after the first year concurrent with declines in redd counts in the NFC. However, effort was inconsistent between years. The mean TL of WCT increased from 301 to 333 mm during the study, and exploitation was relatively low (range = 7 to 18%), indicating sustainable management. However, the reporting rate for WCT tagged during 2015 (15.0%) was also low compared to the statewide mean (49.4%) and the variability between years is unknown. A higher percentage of tags were returned from those tagged by Region 2 staff in the Little North Fork Arm of the reservoir (10.7%) than from those tagged by Region 1 staff in the Little North Fork Clearwater River (0.7%) as part of a separate study, suggesting a collaborative efforts between regions may be a better approach to evaluating exploitation in the Little North Fork Clearwater River. Hybridization was only detected in 20.9% of WCT, and most *O. mykiss* ancestors were derived from local RBT/steelhead populations, rather than coastal origin hatchery trout. Therefore, hybridization with hatchery origin RBT stocked in the reservoir is not a major concern and current fishing rules appear appropriate for the WCT fishery.

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INTRODUCTION

The North Fork Clearwater River (NFC) was the most popular fishing destination for resident fish species in the Clearwater Region, based on the number of angler trips during 2011 (Thomas MacArthur, IDFG, *unpublished data*). It also accounted for approximately 50% of angler trips for resident species in lotic waters. It provides both harvest and catch-and-release opportunities for Westslope Cutthroat Trout (WCT) *Oncorhynchus clarkii lewisi* and Rainbow Trout (RBT) *O. mykiss* depending on the time of year and river section. It also comprises a core area for Bull Trout (BT) *Salvelinus confluentus*, which are listed as Threatened under the Endangered Species Act (ESA).

During the summer, WCT and RBT typically inhabit the NFC and its tributaries. By late May, BT begin migrating out of the reservoir toward spawning habitat in the upper portions of tributaries (Hanson et al. 2014). With cooler water temperatures during autumn months, and completion of spawning for BT, all three species move downstream, entering Dworshak Reservoir during late October and early November. These fish then congregate near slack water interfaces in early November, creating an opportunity to sample them.

The 2019-2024 Fish Management Plan directs IDFG to maintain or increase the size and abundance of WCT, and increase the abundance of naturally produced RBT in the NFC (IDFG 2019). In all tributaries, except Kelly Creek, IDFG is directed to provide harvest opportunity while maintaining size and abundance of both species. In Kelly Creek and its tributaries, IDFG is directed to maximize the size and abundance of trout species. Mangers are directed to protect BT throughout the watershed (IDFG 2019).

In concordance with the Fish Management Plan, fishing season and limits vary by time and location within the watershed. In the NFC, barbless hooks are required, no bait is allowed except for maggots, and no trout harvest is allowed from December 1 through the Friday before Memorial Day weekend. From Memorial Day weekend through November 31, anglers can harvest 2 trout, WCT < 356 mm (14 inches) must be released, and no bait is allowed. There are no size restriction for RBT. However, the general trout limit of 2 per day with no size restrictions applies to all tributaries except Kelly Creek and its tributaries, for which there is no trout harvest or bait allowed at any time.

OBJECTIVES

1. Determine the relative proportions of WCT, RBT, and their hybrids overwintering in Dworshak Reservoir.
2. Assess trends in length distributions of WCT and BT overwintering in Dworshak Reservoir.
3. Estimate annual exploitation of WCT in the North Fork Clearwater Drainage.
4. Determine the degree of hybridization between WCT and RBT, and which strains of RBT are present and have hybridized with WCT.

STUDY AREA

The NFC originates in the Bitterroot Mountains on the border of Idaho and Montana. The watershed drains nearly 632,000 ha within the Clearwater National Forest, which is composed primarily of montane forests in steeply sloped terrain (Falter et al. 1977). The NFC basin can be divided into the NFC in Region 2, and Little North Fork Clearwater River (LNFC) in Region 1. The underlying geology is composed of Columbia River basalt and metamorphic sediments with granitic intrusions covered by shallow soils (Falter et al. 1977). The lower portion of the river was impounded after the construction of Dworshak Dam in 1972, creating Dworshak Reservoir, extending from approximately river kilometer (RKM) 3 to RKM 89, and blocking access for anadromous fish.

METHODS

SALMONID ASSEMBLAGE MONITORING

During the first week of November in 2011, 2015, and 2018, salmonids were captured using angling gear near the slack-water interface of the Little North Fork Arm (LNFA) and North Fork Arm (NFA) of Dworshak Reservoir (Figure 18). Efforts in 2011 included 2 boats and 4 to 5 anglers, and subsequent years included 4 boats and approximately 12 anglers. Fish were measured to the nearest mm total length (TL).

EXPLOITATION OF WILD TROUT

During all three years, WCT > 200 mm TL were tagged using Floy T-bar anchor tags to estimate exploitation. A total of 372 WCT were tagged during all three years. Of these, 82 were tagged in 2011, and 145 were tagged in each 2015 and 2018. Of those tagged in 2011, 52 were tagged in the LNFA, and 30 were tagged in the NFA. In 2015, 28 were tagged in the LNFA, and 117 were tagged in the NFA. In 2018, 33 were tagged in the LNFA, 42 were tagged in the NFA (Figure 22), and location could not be tied to the remaining 70.Tags were subsequently reported by anglers using the IDFG Tag-You're-It program. Exploitation and use were estimated for one year post release following the methods of Meyer and Schill (2014). Reporting rate was estimated using $50 reward tags stocked in 2015, and tag loss was evaluated using double tagged WCT (Meyer et al. 2012).

Reporting rates and tag loss were evaluated using 45 reward tags and 69 double-tagged fish in 2015. Of the 45 reward and 100 non-reward tags stocked in 2015, nine reward and three non-reward tags were reported by anglers, resulting in a reporting rate of 15.0%. Of the 69 fish double tagged in 2015, only one was reported by anglers. Due to the lack of returns from double-tagged fish, the statewide rate for annual tag loss with wild trout was assumed.

HYBRIDIZATION BETWEEN WESTSLOPE CUTTHROAT TROUT AND RAINBOW TROUT

During 2015, fin clips were taken from fish identified in the field as WCT, RBT, or WCTxRBT hybrids (HYB). Fin clips were mounted on Rite-in-the-Rain paper, sent to the genetics lab in Eagle, ID, and assayed at a panel of single nucleotide polymorphisms (SNPs) to assess hybrid status. Genomic DNA was extracted using a 96-well plate extraction protocol following manufacturer's instructions (Nexttec 1-Step DNA Isolation kit, Leverkusen, Germany). Two panels of SNPs were amplified per sample using Genotyping-in-Thousands by sequencing (GT-Seq) protocol developed by Campbell et al. (2015). This method involves using a high-throughput sequencing platform (e.g., Illumina NextSeq 500) to sequence multiplexed PCR products consisting of 50-500 SNPs per individual.

One SNP panel was developed specifically for use with WCT and the other for RBT. Once genotyped, any samples that had an incomplete genetic profile (i.e., data was available for < 90% of the SNP panel per individual) or represented a duplicate genotypes (i.e., individuals that were accidentally sampled twice) were removed prior to downstream analysis. Of the markers in each of the SNP panels, 21 were selected for their ability to differentiate WCT and RBT as they show fixed allelic differences between the two species (e.g., for SNP 1, all RBT samples will have an AA genotype while all WCT will have a GG genotype).

The hybrid status of individual trout was assessed using two complimentary approaches. First, the Bayesian clustering algorithms implemented in the program STRUCTURE (Pritchard et al. 2000) were used to place individuals of unknown ancestry into one of two putative genetic groups (RBT and WCT). This software analyzes differences in the distribution of genetic variants among populations within a Bayesian framework and iteratively places samples into groups with shared, or similar patterns in genetic variation. In STRUCTURE analyses, the user specifies the number of potential populations (K) or assumed genetic groups in the collection of individuals. Each sample is then given a membership coefficient (Q) that is associated with an individual belonging to each of the K potential populations. When genetic admixture is suspected, an individual's membership coefficient (Q) can be attributed to multiple populations (K). This, in essence, identifies the probability that an individual has ancestry belonging to one or more populations. For each individual, the sum of Q values is equal to one. We estimated individual ancestry assuming a *K* of 2 (WCT and RBT) using 10,000 burn-in iterations and 50,000 sampling iterations. Individuals with 50% of their genome attributed each to WCT and RBT would represent an F1 hybrid (i.e., the offspring of a WCT mating with a RBT). Fish with different combinations of the two genomes (e.g., 0.75 WCT and 0.25 RBT) would be hybrids that occurred between an F1 fish and a WCT. Estimates of hybrid class using allele frequencies is only reliable for the first two generations of hybridization (i.e., we cannot reliably differentiate a 3rd or 4th generation hybrid apart; Anderson and Thompson 2002). For sake of simplicity we have classified fish as either parental strain (WCT, RBT), F1, or >F1.

The second analysis used to classify individuals into hybrid classes involved NewHybrids, which places individual fish into six distinct genealogical or hybrid classes (pure WCT, pure RBT, F1 hybrid, WCT backcross, RBT backcross, and F2 hybrid) based on patterns in allele inheritance (Anderson and Thompson 2002). Individuals were assigned to different hybrid classes based on model outputs, which included 10,000 burn-in iterations followed by 20,000 sampling iterations. As with STRUCTURE analysis above, we used outputs from NewHybrids to identify whether hybridization events likely occurred in the distant past (>F1) or if they were continuing to occur presently (F1). For example, F1 individuals in the sample would imply ongoing hybridization, whereas if only WCT or RBT backcrosses (>F1) were present, that would imply historical hybridization (more than one generation ago).

<u>RAINBOW TROUT GENETIC ANCESTRY</u>

An additional suite of analyses were performed to evaluate the genetic ancestry of RBT within the North Fork Clearwater River. Specifically, we were interested in determining whether RBT signatures were of native Redband Trout or non-native coastal origin RBT. Historically, reproductively viable, diploid RBT trout, potentially of coastal origin, were stocked in the North Fork Clearwater River. We compared WCT samples from this study against a suite of reference RBT samples to determine which RBT stocks were most likely present in hybridized individuals.

Whereas the WCT/RBT hybridization analyses above relied on the use of 21 species-diagnostic SNPs, the RBT genetic ancestry results presented here used a marker panel of 242 bi-allelic SNPs and 92 microhaplotypes (see Hargrove et al. 2021 for marker details)

To identify RBT stocks present in the North Fork Clearwater River samples, we first performed a Principle Component Analysis (PCA) on SNP data to explore patterns in genetic variation across populations. While a variety of analytical techniques are currently used to examine population genetic patterns, many of these approaches rely on assumption-laden genetic models which may or may not be valid depending on context (Jombart et al. 2009). Principle Component Analysis represents a multivariate approach with minimal assumptions that attempts to find a few principal components that maximize the amount variation explained among individuals (Reich et al. 2008). Individuals with similar values for a particular principal component can be interpreted as having similar genetic ancestry for that axes. Principle Component Analysis was performed using the statistical package 'adegenet' (Jombart 2008) in R (R Core Team 2020), and results from the first two components were visualized. We included a suite of reference samples to provide context for the genetic ancestry of RBT observed in WCT samples. Specifically, we included representative samples from six coastal RBT stocks (Arlee, Eagle Lake, Erwin x Arlee, Fish Lake, McConaughy, and Shasta hatchery strains), broodstock from the 1969 spawn year of Dworshak National Fish Hatchery (representing pre-impoundment steelhead from the North Fork Clearwater River) and wild steelhead collections from the lower Clearwater River, South Fork Clearwater River, and upper Clearwater River reporting groups (see Hargrove et al. 2021 for specifics on Clearwater River samples).

In addition to PCA, we performed genetic clustering, as described above (see details above about STRUCTURE). We assumed five potential populations (K = 5) to account for known genetic structure in reference populations, and estimated individual ancestry based on 100,000 burn-in iterations followed by 100,000 sampling iterations. The above model was replicated three times.

RESULTS

SALMONID ASSEMBLAGE MONITORING

During the study, 589 salmonids were captured, including 430 WCT, 82 BT, 28 RBT, and 49 HYB (Table 8). Of these, 168 were captured in 2011, including 105 WCT (63% of the catch), 41 BT (24% of the catch), 8 RBT (5% of the catch), and 14 HYB (8% of the catch; Figure 19). Another 232 were captured in 2015, including 172 WCT (74% of the catch), 22 BT (9% of the catch), 11 RBT (5% of the catch), and 27 HYB (12% of the catch). In 2018, 189 were captured, including 153 WCT (81% of the catch), 19 BT (10% of the catch), 9 RBT (5% of the catch), and 8 HYB (4% of the catch).

Length data was recorded for all salmonids except a single BT. The mean TL (± 95% CIs) of WCT steadily increased from 306 mm (± 9) in 2011, to 318 mm (± 7) in 2015, and 333 mm (± 7) in 2018 (Figure 20). This increase in mean length coincided with a shift toward proportionately more WCT > 300 mm TL in the catch during 2018 (Figure 21). The mean TL of BLT was 429 mm (± 30) in 2011, 442 mm (± 41) in 2015, and 438 mm (± 46) in 2018. However, confidence intervals overlapped for all three years. The mean TL of RBT was 345 mm (± 31) in 2011, 336 mm (± 39) in 2015, and 376 mm (± 48) in 2018. As with BLT, confidence intervals overlapped for all three years. The mean TL of HYB was 327 mm (± 36) in 2011, 336 mm (± 39) in 2015, and 364 mm (± 28) in 2018. Once again, confidence intervals overlapped for all three years.

EXPLOITATION OF WILD TROUT

A total of 33 tags were reported through "Tag, You're It" by the end of 2020. Of these, 13 were from WCT tagged in 2011. Another 10 were reported from both 2015 and 2018. Of those tagged in the LNFA, one was recovered in the reservoir and 13 were recovered in in free-flowing water (12 in LNFC, and 1 in Breakfast Creek; Figure 22). Of those tagged in the NFA, one was recovered in the reservoir and 12 were recovered in free-flowing water (11 in the NFC and 1 in the LNFC; Figure 21). Precise tagging location was not available for 6 tag returns, 2 recovered in the LNFC and 4 recovered in the NFC.

Of the 33 tags reported, 26 were caught within 365 days of tagging and were used to estimate exploitation and use (Table 9). Exploitation for all sizes was 18% in 2012, 7% in 2016, and 10% in 2019. Harvest within the first year for WCT < 365 mm TL during tagging occurred in areas without size restrictions (i.e., LNFC and NFC tributaries besides Kelly Creek). Use for all sizes during each of these years was 81%, 15%, and 41%, respectively.

HYBRIDIZATION BETWEEN WESTSLOPE CUTTHROAT TROUT AND RAINBOW TROUT

Of 174 samples processed for inventory, 96 samples were retained for hybrid analysis as they displayed >90% amplification success. Bayesian clustering analysis identified 70 individuals as WCT, twelve individuals as F1 hybrids (i.e., the offspring of a WCT mating with a RBT), eight individuals as advanced stage backcross (i.e., > F1 hybrids). Six individuals that had > 95% (but not 100%) of their genome attributed to WCT ancestry (Figure 23).

Estimation of genetic ancestry based on allele frequencies as calculated in New Hybrids produced results that were highly congruent with Bayesian clustering analysis. Specifically, 86 of the 96 samples analyzed via both methods were placed into the same hybrid class and the 10 discrepancies fell into 1 of 2 classes. In four cases, STRUCTURE identified an individual as a >F1 hybrid whereas New Hybrids identified the same individuals as pure WCT. In an additional six cases, New Hybrids classified an individual as pure WCT whereas Structure identified >95% of these fish genomes were attributed to WCT.

Using two complimentary techniques, we show that hybridization between WCT and RBT has both occurred historically and continues to occur in the North Fork Clearwater River, but that 79.1% of fish are likely pure WCT. The large proportion of fish that failed to genotype was surprising as we attempted to amplify samples that initially failed an additional two times. This type of result is generally associated with poor DNA preservation. Alternatively, if these fish were small and phenotypically ambiguous (i.e. potentially a species other than WCT or RBT), amplification could have failed due to a lack of compatibility with our marker panels.

RAINBOW TROUT GENETIC ANCESTRY

Combined, the first and second components of the PCA explained 22.5% and 5.8% of the observed variance in genetic diversity (Figure 24). We observed three distinct clusters in the PCA output, with only a handful of individuals from different clusters overlapping. The first cluster consisted of Dworshak 1969 broodstock samples and steelhead collections from the lower, South Fork, and upper Clearwater River reporting groups. The second cluster contained coastal RBT hatchery stocks which did not exhibit overlap with any other sample collections. The third cluster contained WCT samples from the North Fork Clearwater, some of which overlapped with reference collections of wild steelhead from the Clearwater River reference collections, suggesting shared allelic ancestry. The absence of overlap between WCT and coastal RBT samples supports the claim that coastal RBT ancestry is minimal, if any, among these WCT samples from the North Fork Clearwater River.

Results from genetic clustering were similar to those from our PCA and hybrid analyses above. Namely, the majority of WCT samples belonged to a genetic cluster not shared with any RBT samples (i.e., WCT; Figure 25). Second, individuals with genetic admixture (consisted of more than one genetic cluster) generally had ~50% of their genome attributed to a non-WCT cluster. This observation is what we would expect of F1 hybrids, which is how many of these same samples were classified based upon hybrid analysis. Lastly, admixed WCT were generally a combination of the WCT genetic cluster and a Clearwater River reference collection genetic clusters (i.e., Lower Clearwater or Dworshak/South Fork Clearwater). Only one WCT sample had appreciable coastal RBT ancestry (<25%), suggesting coastal RBT influence among the surveyed WCT samples was likely minimal.

DISCUSSION

The current seasons and limits structure seem to be adequate to conserve populations of WCT that overwinter in Dworshak Reservoir. While the number of WCT captured increased from the initial year of the study, effort was not consistent. Therefore, numbers of WCT caught during this study may not reflect changes in abundance. However, snorkel surveys conducted in the NFC indicate that WCT densities increased in the watershed from 2011 to 2015 (Hand et al. 2019), and densities generally increased in the LNFC through 2015 (Ryan et al. 2018), both of which are consistent with increasing abundance and catches in our angling surveys. No snorkel data were available for 2018.

The mean TL of WCT increased over the course of this study and coincided with a shift to proportionately more WCT > 300 mm TL in 2018. An increase in mean TL may indicate either positive (e.g., increased growth) or negative (e.g., fewer young fish) change to a population. Additional data regarding age specific abundance and growth is needed to determine why this population is shifting toward larger fish and what the long-term consequences of that shift might be. The length distributions of WCT captured in the reservoir also differed from the WCT observed during snorkel surveys in the NFC (Hand et al. 2020) and LNFC (Ryan et al. 2018) in that smaller WCT observed in the snorkel surveys were absent from the reservoir survey. However, lengths of WCT captured in the reservoir during 2015 were similar to those captured during angling surveys in the LNFC the same year (Ryan et al. 2018). Therefore, the WCT sampled during this study are likely representative of the size range encountered by anglers, rather than the population as a whole, and it cannot be ascertained from this study whether smaller WCT overwinter in the reservoir. However, the observed increase in TL over time is consistent with low exploitation and sustainable fishing rules.

The number of BT captured in the reservoir declined from the first year despite increased fishing effort during subsequent years (data not shown). The decline in BT catch corresponds with declines in redd counts (Hand et al. 2020), and therefore may be indicative of an actual population decline. In the future, fishing effort will be tracked in order to use catch per unit effort as an index of abundance.

Exploitation of WCT tagged in the reservoir (during our study) was low, further suggesting that current fishing rules are sufficient to sustain the quality of the fishery. Exploitation for WCT tagged in the reservoir was generally higher than the estimate derived from WCT tagged in the LNFC during 2016 (2%; Ryan et al. 2018). We did not attempt to derive an exploitation estimate for WCT tagged in the LNFA due to small sample sizes. However, more WCT were reported being caught in the LNFC one year after release (three returns) than those tagged in the river the earlier that year (one return; Ryan et al. 2018). This occurred despite a larger release group in the river during July 2015 (134 tags) compared to the number tagged in the LNFA during November of that year (28 tags). This suggests that tagging efforts in the reservoir should be coordinated with efforts of Region 1 staff tagging in the river to provide a broader picture of the fishing effort in the LNFC. While WCT were tagged in the reservoir during this study, the majority (> 90%; Figure 22) were caught in free flowing water above the reservoir. Estimates of use were much higher than exploitation, indicating that many fish are released and that these fish provide significant angling opportunities, even in areas without size restrictions (i.e., LNFC and tributaries of the NFC other than Kelly Creek).

Reporting rates for this study were low compared to those reported for wild trout by Meyer et al. (2012). It is not known whether the actual reporting rates were consistent for all years, since reward tags were only used in 2016. Estimates of use varied widely between years, which could have been due in part to undetected changes in reporting rates (Pollock et al. 2001). Changes in reporting rates are often consistent over periods of several years (Green et al. 1983; Parsons and Reed 1998), but studies examining reporting rates over the period of this study are difficult to find. In other studies, angler reporting rates can vary by broad geographical region in similar fisheries (Denson et al. 2002). Due to the low return rate estimated for 2016, and used for all years of this study, reward tags should be used in future years to assess the consistency, or lack thereof, for return rates.

Hybridization between WCT and RBT has been a concern to fisheries managers, particularly where hatchery RBT are stocked in waters with native WCT (Weigel et al. 2003; McKelvey et al. 2016). Hybridization rates in this study (79% pure WCT) were higher than what is typically acceptable (< 10% non-native species alleles), and at the upper end of acceptability for extenuating circumstances (< 20%; IDFG 2013). However, samples for this study were only collected in the furthest downstream reaches for this population. Weigel et al. (2003) found that the occurrence of hybridization decreased with both elevation and stream width, and that sampling for hybridization should be sufficiently spread throughout the study stream. While sampling for this study was spatially limited, the fish were also migratory, with tags recovered throughout the watershed. Still, this estimate of hybridization should be viewed with caution and a spatially broader study should be conducted to confirm these results.

Although the level of hybridization detected during this study may be of concern, there was very little evidence of hybridization with non-native RBT (i.e., coastal origin RBT from hatchery stocking). Almost all the hybridization detected in this study occurred with native RBT or remnants of steelhead that once occurred in the North Fork Clearwater River, whereas previous studies assumed hybridization was the result of non-native (i.e., hatchery origin) strains of RBT (Weigel et al. 2003). While this level of hybridization is higher than previously observed in sympatric populations of native WCT and RBT in Idaho, hybridization between sympatric species can occur naturally and may not have the same conservation concerns as hybridization caused or exacerbated by anthropogenic causes (Kozfkay et al. 2007). Weigel et al. (2003) also concluded that factors such as habitat preference, mate selection, and survival of hybrids were more important than stocking in determining the distribution of hybrids. Further study is needed to determine the true level, nature, and potential threat of hybridization between native RBT and

WCT in the NFC. However, these results suggest that current stocking of hatchery RBT in Dworshak Reservoir is not genetically detrimental to WCT overwintering in the reservoir.

MANAGEMENT RECOMMENDATIONS

1. Maintain current trout regulations for Dworshak Reservoir and the North Fork Clearwater River.

2. Continue to monitor length distributions of WCT, BT, RBT and HYB that overwinter in Dworshak Reservoir.

3. Monitor CPUE during future surveys as an index of the abundance of WCT, BT, RBT and HYB that overwinter in Dworshak Reservoir.

4. Continue to monitor exploitation of WCT that overwinter in Dworshak Reservoir and use reward tags to accurately estimate tag reporting rates.

5. Replicate the broader genetics study conducted by Weigel et al. (2003) to re-evaluate WCT hybridization in the Clearwater River drainage.

Table 8. Numbers of salmonids captured by angling in the Little North Fork Clearwater Arm (LNFA), North Fork Clearwater Arm (NFA), or location not recorded (LNR) of Dworshak Reservoir during each of three years. Species include Bull Trout (BLT), Westslope Cutthroat Trout (WCT), Rainbow Trout (RBT), and RBTxWCT hybrids (HYB).

Species	Year	LNFA	NFA	LNR	Total
BLT	2011	30	11	0	41
	2015	3	9	10	22
	2018	0	0	19	19
	All years	33	20	29	82
WCT	2011	64	41	0	105
	2015	28	130	14	172
	2018	33	42	78	153
	All years	125	213	92	430
RBT	2011	4	4	0	8
	2015	0	5	6	11
	2018	0	0	9	9
	All years	4	9	15	28
HYB	2011	1	13	0	14
	2015	0	16	11	27
	2018	0	0	8	8
	All years	1	29	19	49
All specie	All years	163	271	155	589

Table 9. Numbers of Westslope Cutthroat Trout captured by angling in Dworshak Reservoir and tagged using t-bar anchor tags, along with number reported and harvested. Summaries are reported for the Little North Fork Clearwater Arm (LNFA), North Fork Clearwater Arm (NFA), or location not recorded (LNR), and for WCT < 356 mm (14 inches) TL and ≥ 356 mm TL.

Year	Size	Disposition	LNFA	NFA	LNR	Total
2011	< 356 mm TL	caught	44	35	0	79
		tagged	36	24	0	60
		reported	8	1	0	9
		harvested	2	0	0	2
	≥ 356 mm TL	caught	20	6	0	26
		tagged	16	6	0	22
		reported	0	0	0	0
		harvested	0	0	0	0
	All sizes	caught	64	41	0	105
		tagged	52	30	0	82
		reported	8	1	0	9
		harvested	2	0	0	2
2015	> 356 mm TL	caught	21	101	14	136
		tagged	21	89	0	110
		reported	3	4	0	7
		harvested	0	2 [1]	0	2 [1]
	≥ 356 mm TL	caught	7	29	0	36
		tagged	7	28	0	35
		reported	0	2	0	2
		harvested	0	1 [1]	0	1 [1]
	All sizes	caught	28	130	14	172
		tagged	28	117	0	145
		reported	3	6	0	9
		harvested	0	3 [2]	0	3 [2]
2018	> 356 mm TL	caught	14	32	49	95
		tagged	14	32	44	90
		reported	1	1	3	5
		harvested	0	1	0	1
	≥ 356 mm TL	caught	19	10	29	58
		tagged	19	10	26	55
		reported	0	1	2	3
		harvested	0	1	1 [1]	2
	All sizes	caught	33	42	78	153
		tagged	33	42	70	145
		reported	1	2	5	8
		harvested	0	2	1 [1]	3 [1]
All years combined		caught	125	213	92	430
		tagged	113	189	70	372
		reported	12	9	5	26
		harvested	2	5 [2]	1 [1]	8 [3]

[1]One WCT was reported to have been harvested only because it was tagged.
[2]Two WCT were reported to have been harvested only because they were tagged.
[3]Three WCT were reported to have been harvested only because they were tagged.

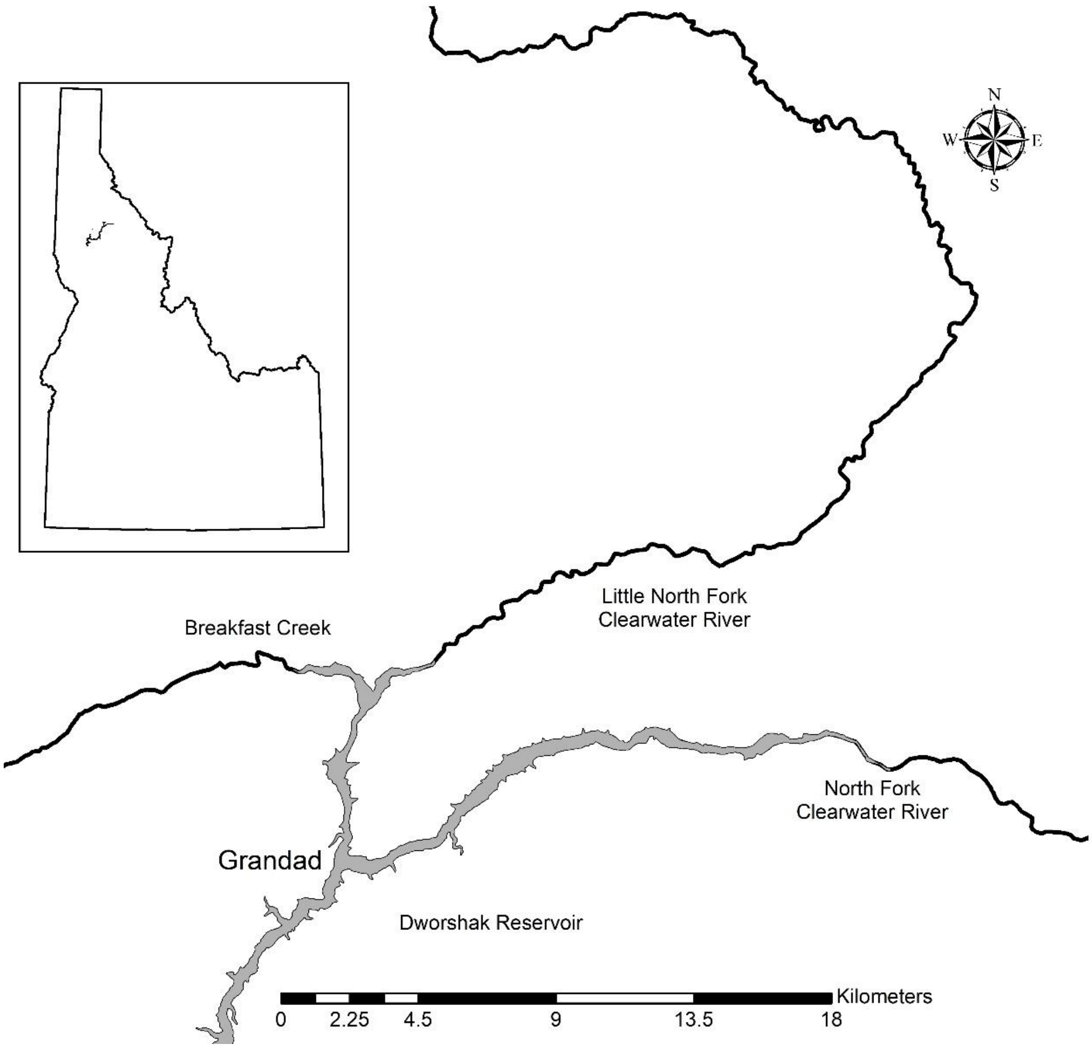

Figure 18. Map of the upper end of Dworshak Reservoir, including arms formed by the inundation of the Little North Fork Clearwater River (Little North Fork Arm, LNFA) and North Fork Clearwater River (North Fork Arm, NFA) and the location within the state of Idaho.

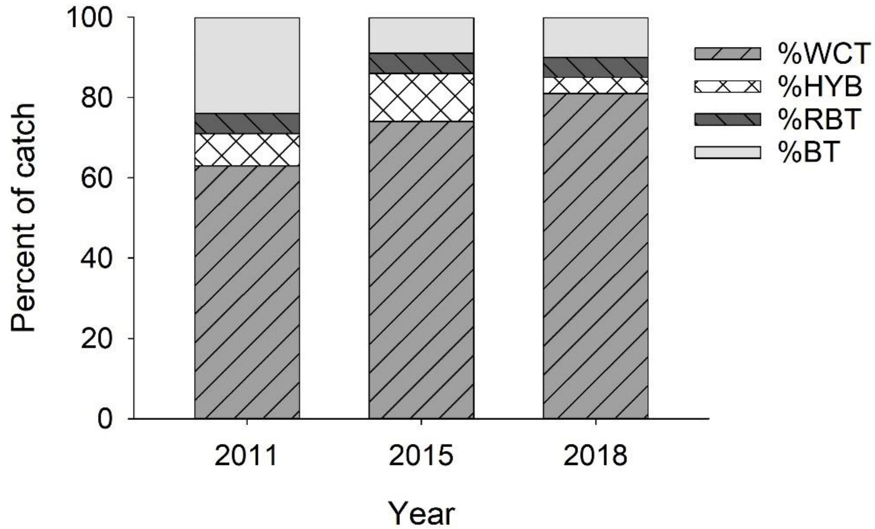

Figure 19. Catch composition of salmonids captured by angling in Dworshak Reservoir by year of capture. Species include Westslope Cutthroat Trout (WCT), Rainbow Trout (RBT), WCTxRBT hybrids (HYB), and Bull Trout (BT). Sample sizes for each year were 168, 232, and 189 fish.

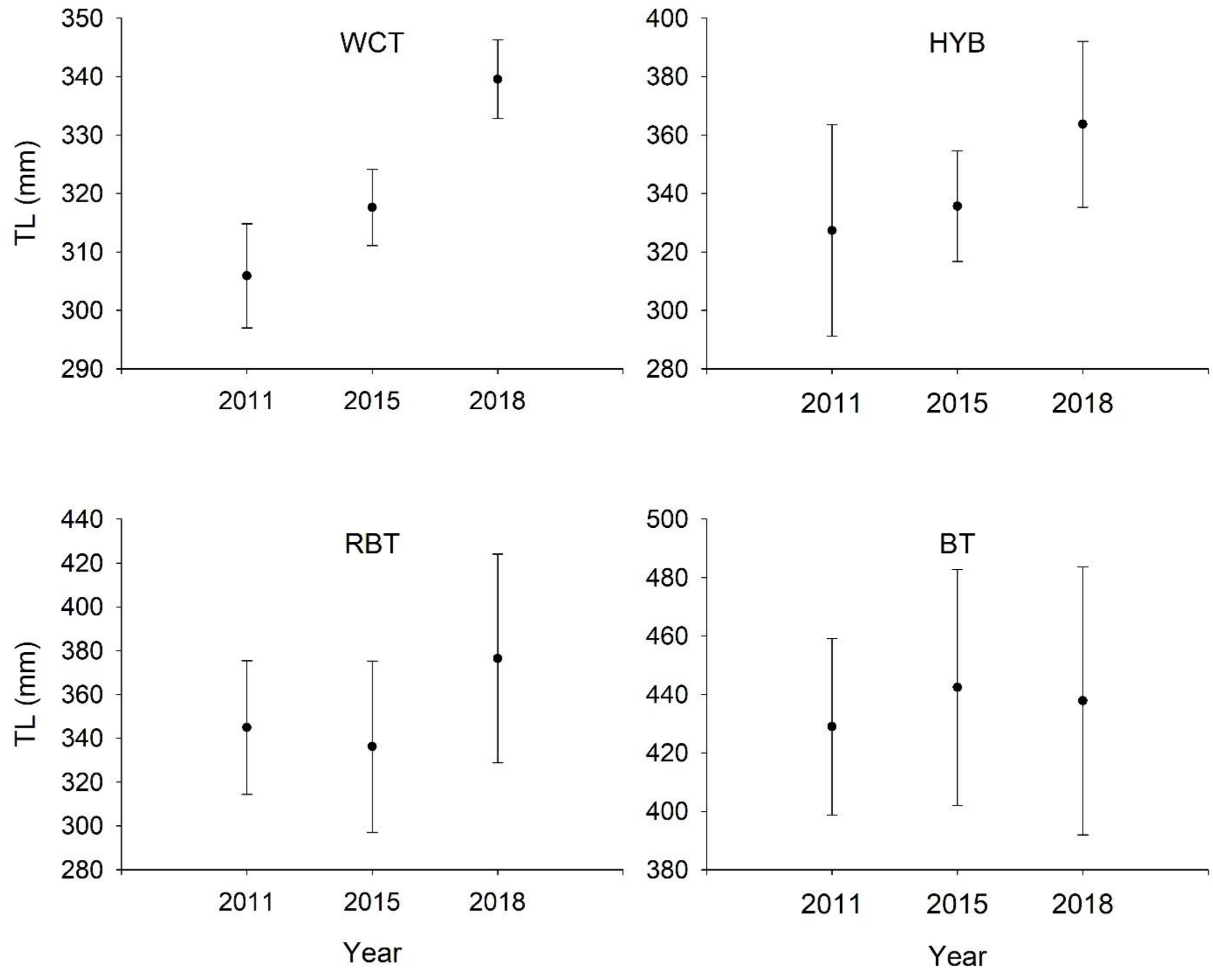

Figure 20. Mean lengths of Westslope Cutthroat Trout (WCT), Rainbow Trout (RBT), WCTxRBT hybrids (HYB), and Bull Trout (BT) collected by angling from Dworshak Reservoir. Sample sizes by year for each species were 105, 172, and 153 for WCT; 14, 27, and 8 for HYB; 8, 11, and 9 for RBT; and 40, 22, and 19 for BT. Error bars represent 95% confidence intervals.

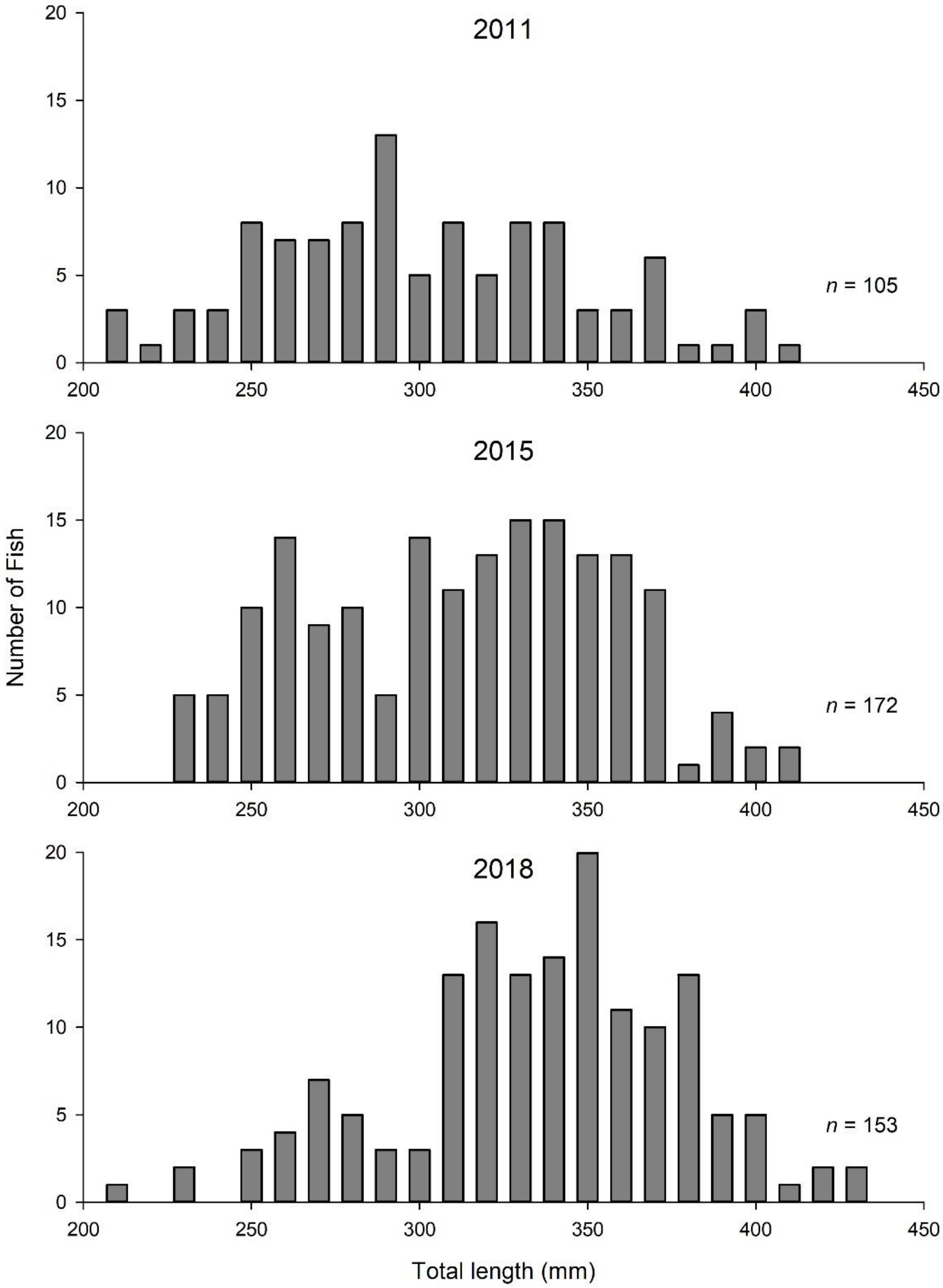

Figure 21. Length-frequency distributions of Westslope Cutthroat Trout by year captured by angling near the slack-water interface on Dworshak Reservoir.

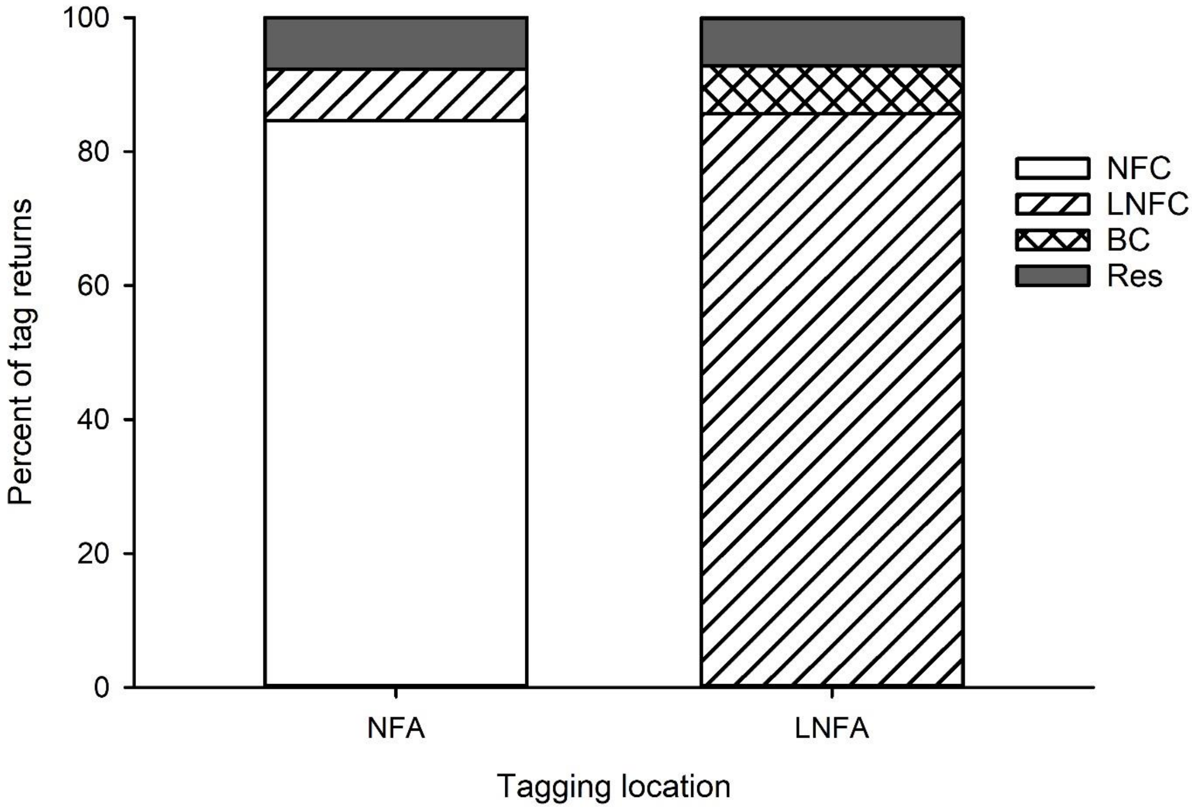

Figure 22. The proportions of Westslope Cutthroat Trout tagged in the North Fork Arm (NFA, n = 13) and Little North Fork Arm (LNFA, n = 14) of Dworshak Reservoir, that were reported as being caught in the free-flowing North Fork Clearwater River (NFC), Little North Fork Clearwater River (LNFC), Breakfast Creek (BC), or Dworshak Reservoir (Res).

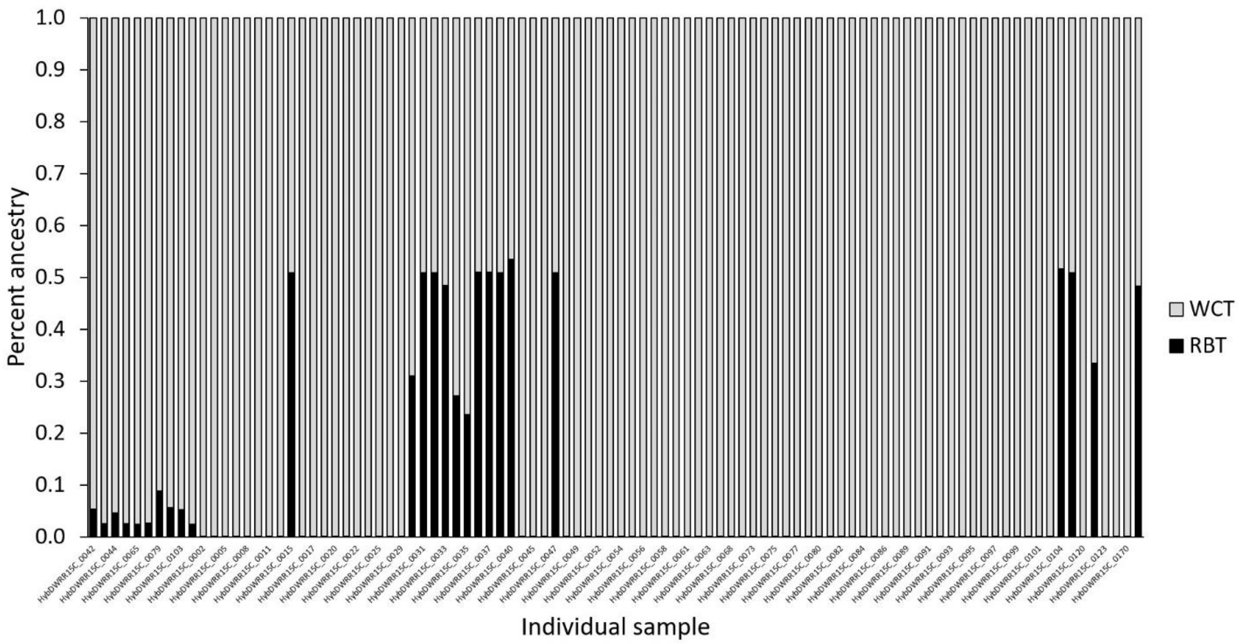

Figure 23. The genetic ancestry of individual fish sampled from the North Fork Clearwater River based on 21 diagnostic SNP markers. Bars correspond to individual fish and colors correspond to the proportion of a fishes genome attributed to Westslope Cutthroat Trout (WCT; grey) and Rainbow Trout/steelhead (RBT; black).

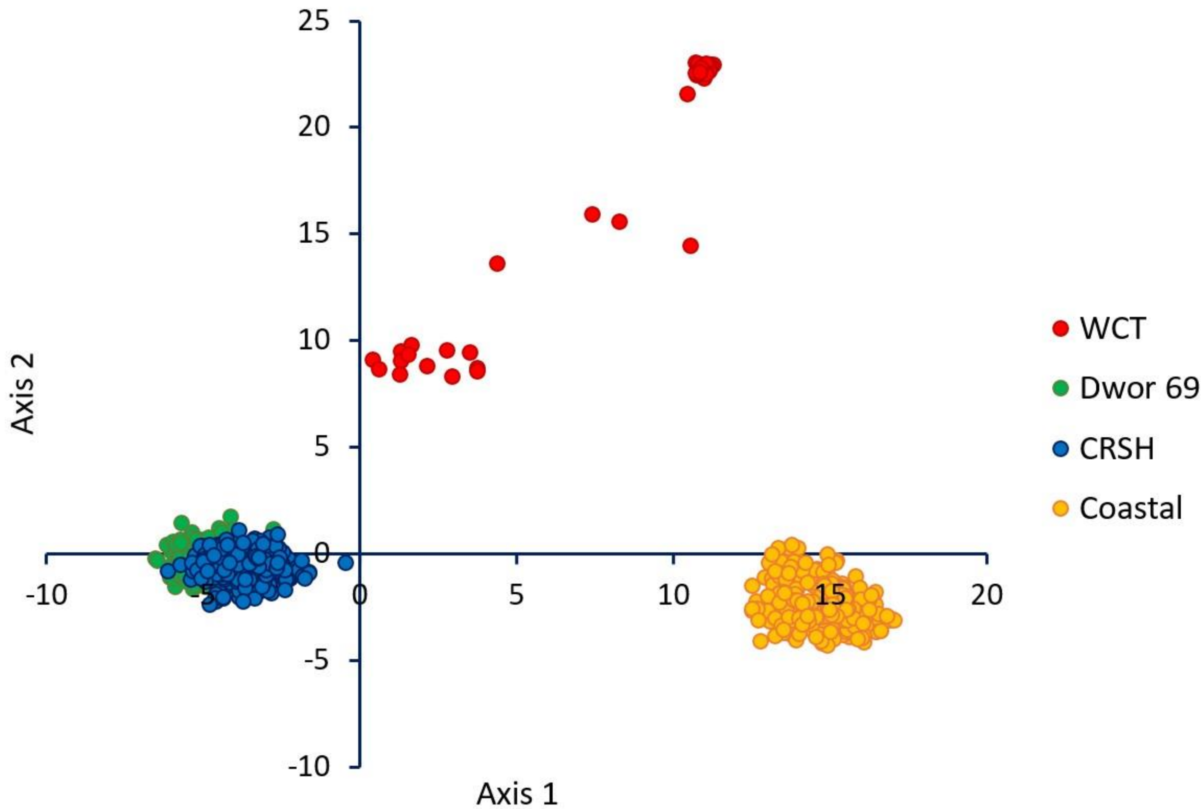

Figure 24. Principal component analysis of Westslope Cutthroat Trout (WCT; red circles) sampled from the North Fork Clearwater River along with reference collections of Rainbow Trout/steelhead collections from Dworshak National Fish Hatchery broodstock spawn year 1969 (Dwor 69; green circles), and baseline samples from Clearwater River steelhead (CRSH; blue circles), and Rainbow Trout from coastal hatcheries (Coastal; yellow circles). Plot labels are placed on the population centroid (mean value across all individuals within a population) and circles correspond to individuals.

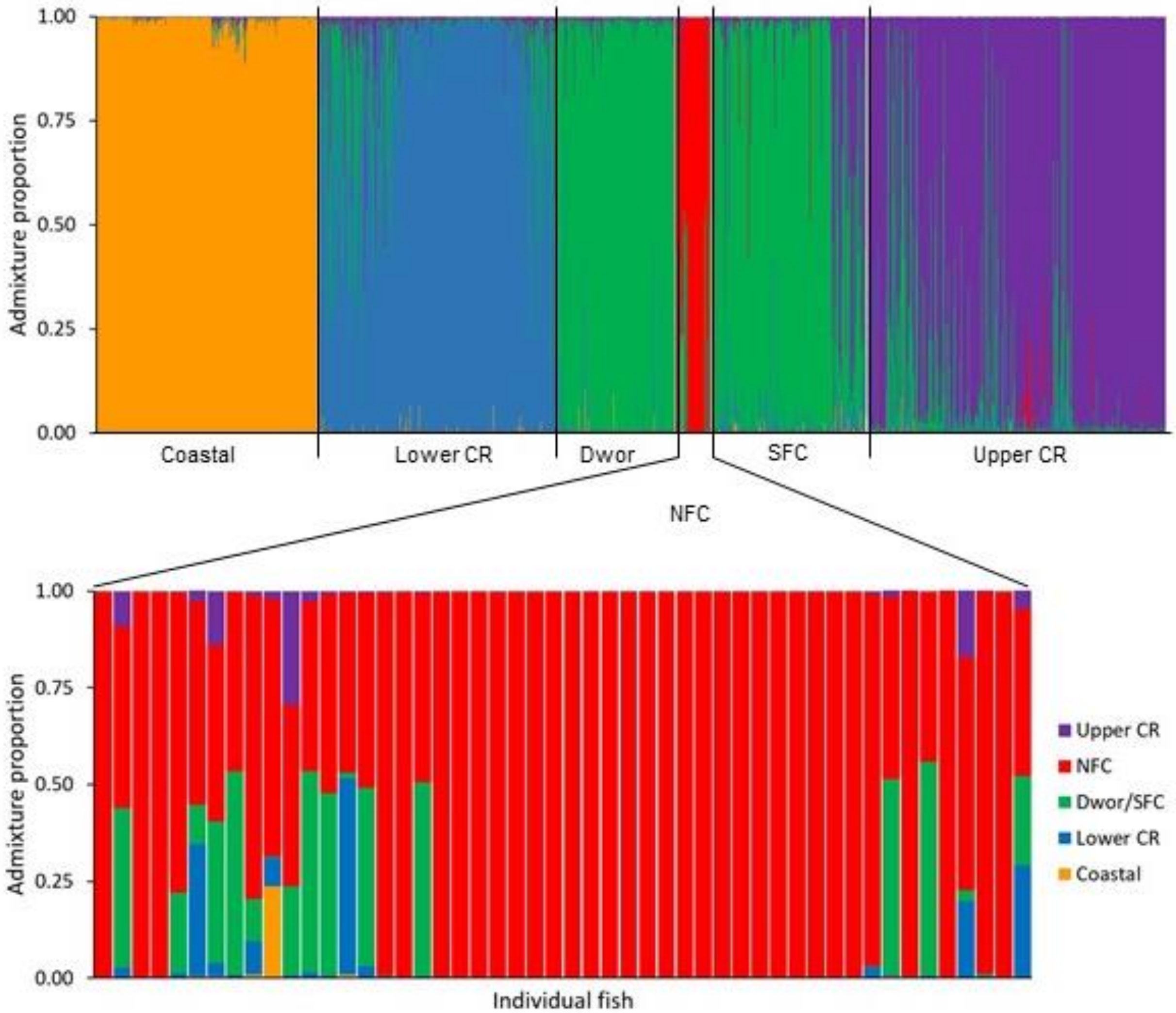

Figure 25. A barplot displaying admixture proportions assuming a K = 5 (number of populations) for Westslope Cutthroat Trout (WCT; red) and Rainbow Trout/steelhead collections from coastal hatcheries (Coastal; yellow), Dworshak National Fish Hatchery broodstock spawn year 1969 and South Fork Clearwater steelhead (Dwor/SFC; green), and the upper Clearwater River (Upper CR; purple), and lower Clearwater River (Lower CR: blue). Vertical bars represent an individual fish and the colors represent distinct genetic clusters (K) identified by STRUCTURE.

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EVALUATION OF FISH POPULATIONS IN THE LOCHSA RIVER

ABSTRACT

A snorkel survey was conducted in the Lochsa River drainage in 2018, adding to a long-term dataset to assess trends in Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT), Rainbow Trout *O. mykiss* (RBT), and Mountain Whitefish *Prosopium williamsoni* (MWF) abundance and size distribution and to evaluate whether fishing regulations were sufficient to maintain a high abundance of larger WCT. Mean linear density of WCT for all main-stem Lochsa River transects increased steadily from 1975 to 1981 and then declined in 2017 and 2018. Westslope Cutthroat Trout were observed in 82% of modern transects (since 2013), with the highest areal densities occurring in Colt Killed Creek. The proportion of WCT observed in 2018 that were > 305 mm in total length for all transects in the Lochsa River drainage was 62%, the highest observed in the modern surveys. Rainbow Trout were observed in 68% of transects, and mean linear density increased in all three survey sections compared to 2013 and 2017. In 2018, RBT > 305 mm in total length accounted for 14% of those observed in the Lochsa River drainage. Mean areal density of MWF in all transects was the same as 2017, but was only about 25% of that observed in 2013. One adult Smallmouth Bass *Micropterus dolomieu* was observed during our sampling in the Lochsa River for the first time. The historic increases in WCT density have been attributed to the implementation of catch-and-release regulations in 1977. In contrast, the recent declines are likely attributable to environmental conditions such as low flow and higher temperatures that could have reduced fish density in the Lochsa River through increased mortality or influenced migration upstream out of survey areas to seek colder water. For RBT, long-term trends in density were probably affected by cessation of hatchery stockings in 1990. Recent declines in steelhead smolt out-migration and adult returns throughout the Clearwater River basin are also likely influencing densities observed in modern surveys. However, wild steelhead adult returns were higher in the 1970s than 2010 - 2016. Thus, while lower returns are likely impacting current densities, it would not explain the decline observed in modern surveys. While our data in the Lochsa River is short-term, declines in MWF densities have been observed in the main-stem of other northern Idaho rivers. The direct cause of these declines has not been identified, but they have been linked to occurrences of low flows and higher water temperatures. The observation of one adult Smallmouth Bass indicates that they are moving into the Lochsa River basin, but have not likely reproduced in this system.

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INTRODUCTION

Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT) are distributed throughout the Lochsa River drainage, occupying the main-stem river and tributaries. Both resident and fluvial life history forms are present. US Highway 12, which runs along the Lochsa River, was completed in 1962. Its completion opened up the entire length of the Lochsa River to easy access for anglers. By 1966, the WCT population was considered to have been drastically reduced, likely due to high levels of harvest (Mallet 1967; Dunn 1968; Rankel 1971; Lindland 1977). A 1956 creel survey estimated WCT catch at 5,948 fish (Corning 1956). By 1976, creel surveys showed catch had dwindled to 654 WCT (Lindland 1977). The decline of this WCT population prompted the implementation of catch-and-release regulations upstream of the Wilderness Gateway Campground bridge in 1977.

The Lochsa River watershed also supports wild runs of spring and summer Chinook salmon *O. tshawytscha*, summer steelhead, and Pacific Lamprey *Entosphenus tridentatus*. Additionally, hatchery releases of summer Chinook salmon occur in this watershed. Resident Rainbow Trout, Bull Trout *Salvelinus confluentus* (BT), and Mountain Whitefish *Prosopium williamsoni* (MWF) also occur in the watershed. Bull Trout are located mainly in the upper main-stem Lochsa River and the higher elevation streams, and MWF occur primarily in the main-stem river and the largest tributaries. Currently, the management strategy for resident fish in the Lochsa River basin is to maintain a high density of larger WCT and RBT, promote MWF fishing, and maintain a catch-and-release BT fishery (IDFG 2019).

Snorkel surveys have been conducted in the Lochsa River to monitor the WCT population since 1975. Densities increased seven-fold in the catch-and-release section, and four-fold in the harvest section from 1977 to 1981 after the catch-and-release regulations were implemented (Lindland 1982). Although occasional snorkeling surveys have been conducted in the Lochsa River since 1981, the survey in 2013 marked the first occasion since 1981 where the trend surveys established by Graham (1977) were revisited, thus allowing for a direct comparison of observed densities across time (Hand et al. 2016). The primary objective of the 2013 survey was to re-establish trend monitoring to evaluate current WCT densities, while simultaneously establishing trend and presence/absence surveys for other resident fishes, especially Smallmouth Bass *Micropterus dolomieu* (SMB), which were believed to have migrated into the lower Lochsa and Selway rivers.

The Lochsa River provides a popular trout fishery that is often compared to other premier Idaho trout fisheries. Trout fishing in the Lochsa River is managed with limited restrictive harvest regulations and catch-and-release fishing. In 2017, the Lochsa River drainage had three harvest management strategies for trout to provide a diversity of opportunity for anglers and to maintain a high density of larger WCT. For the main-stem Locsha River from the mouth to the Wilderness Gateway Bridge, the daily limit was two trout with none under 356 mm from Memorial Day weekend through November 30. For the main-stem Lochsa River from the Wilderness Gateway Bridge (WGB) to the confluence of Colt Killed and Crooked Fork creeks and Crooked Fork from its mouth to Brushy Fork, the trout season and limit was catch and release, open all year. Crooked Fork Creek upstream of Brushy Fork Creek and all other tributaries of the Lochsa River were managed under the Clearwater Region general rules (harvest of two trout, any size, open all year). As demand on these fisheries continues, it is important to track the status of fish populations to ensure continued quality fishing and to conserve wild native trout populations.

OBJECTIVES

1. Assess trends in density of Westslope Cutthroat Trout and whether a high density of larger fish is being maintained in the Lochsa River.

2. Evaluate trends in density and size structure of Rainbow Trout *Oncorhynchus mykiss* (RBT; resident RBT and juvenile steelhead) and Mountain Whitefish *Prosopium williamsoni* in the Lochsa River.

3. Evaluate whether the distribution of Smallmouth Bass is increasing in the Lochsa River.

STUDY AREA

The headwaters of the Lochsa River drainage are found in the Bitterroot Mountains on the Idaho-Montana border (Figure 26). The Lochsa River is formed by the confluence of Crooked Fork Creek and Colt Killed Creek (formerly White Sands Creek) and flows 113 km southwest, joining the Selway River at the town of Lowell, ID, to form the Middle Fork Clearwater River. The Lochsa River drainage covers 3,056 km^2, all in Idaho County. The majority of the watershed occurs at elevations > 1,200 m. Most of the sub-basin is granitic rock that is part of the Idaho batholith. Land ownership in the Lochsa River drainage is mixed, with the majority of the land under public ownership managed by the U.S. Forest Service. Nearly 80% of the drainage is designated as wilderness (Selway Bitteroot Wilderness Area) or roadless. The Lochsa River is designated a Wild and Scenic River. The primary private landowner in the drainage is Western Pacific Timber Company. They, and previous owners, have intensively managed this area for timber production. These actions are believed to impact fish populations in some areas through sedimentation, poor in-stream cover, and impacts from upland disturbances.

METHODS

A snorkel survey was conducted in the Lochsa River basin from July 13 to 17, 2018 to monitor the density and size structure of WCT, Rainbow Trout/steelhead (RBT), and MWF. We surveyed a total of 38 transects in the main-stem Lochsa River, Crooked Fork Creek and Colt Killed Creek (Figure 26). Detailed transect descriptions and locations are provided in Appendix A of Hand et al. (2016). Snorkel surveys were conducted by one or two snorkelers, depending on the width of each transect. A single snorkeler was used only when the entire wetted width of the stream could be effectively observed by one person. The number of snorkelers surveying each transect was consistent with previous surveys to allow for direct comparison of data. Transects were snorkeled downstream, with each surveyor swimming close enough to shore to see the shoreline. Each snorkeler surveyed towards the thalweg and towards their respective shorelines. All fish observed were counted, and total length (TL) was estimated to the nearest inch for most species. Other species (e.g. *Cottus* spp, *Catostomus* spp.) were categorized as > or < 305 mm. Transect length (m) and average width (m; based on five measurements) was measured using a Nikon ProStaff S laser rangefinder. Visibility (m) was estimated at each transect by holding a Keson 50-m, reel-style, fiberglass measuring tape underwater. A snorkeler backed away from the reel until lettering was indistinguishable, then moved back towards the reel until the lettering was viewable again. This distance from snorkeler to the reel was recorded. Habitat type, date, time of day, water temperature, and weather conditions were also recorded for each transect. Juvenile steelhead and resident Rainbow Trout are indistinguishable, and are collectively referred to as "RBT" in this chapter.

We evaluated trends in WCT and RBT density following methods from historic surveys in the Lochsa River (1975 - 1981). This allowed us to compare more recent data (since 2013) to the historic data. Historic surveys did not include collection of MWF data. The historic surveys doubled the transect length to account for when two snorkelers were used and calculated linear density as fish/100 m (Graham 1977; Mabbott 1980; Lindland and Pettit 1981). In this report we refer to these calculations as "linear density". The historic surveys only occurred in the main-stem Lochsa River, and transects were broken into survey sections to help assess which river reaches were used most by fish (during the summer) and how different fishing regulations may be influencing fish linear density and size. The survey sections were delineated as follows: mouth of Lochsa River to Fish Creek, Fish Creek to Lake Creek, and Lake Creek to Crooked Fork Creek. Mean linear density for each survey section and each year were calculated as the means of ratios. We evaluated long-term trends in mean linear density using least squares regression with survey year (1975 - 2018) as the independent variable and $\log_e$ transformed mean linear density as the dependent variable (Maxell 1999; Kennedy and Meyer 2015). The intrinsic rate of change in the population (r_{intr}) was determined by the slope of the regression line fit to these data. A 90% CI was calculated for r_{intr} to determine significance. The trend is considered significant when $r_{intr} \neq 0$ and the error bounds do not include 0. We used a significance level of $\alpha = 0.10$.

We also evaluated trends in WCT, RBT, and MWF density, and density of just those fishes > 305 mm, using area measurements to be consistent with modern estimates for snorkel surveys conducted in Idaho and allow for comparisons to other rivers (Apperson et al. 2015). We did not evaluate fish > 305 mm for historic surveys, as length data was not collected. These densities were calculated as "fish/100 m^2", and were compared to previous surveys conducted in 2013 and 2017 as higher or lower due to small sample size. Densities for survey sections and years were calculated as means of ratios. In this report we refer to these calculations as "areal density". Main-stem Lochsa River survey sections were delineated as follows: mouth of Lochsa River to WGB, WGB to Lake Cr., and Lake Cr. to Crooked Fork Cr. We used different survey sections than were used in historic surveys to delineate among different harvest regulation areas. Density was also compared between years for Crooked Fork Cr. and Colt Killed Cr. Mean density was calculated by averaging individual transect densities to maintain consistency with other Idaho Department of Fish and Game snorkel projects (Apperson et al. 2015). A more thorough statistical analysis of population trends using area density will be conducted after additional surveys have been completed. Distribution and densities of WCT, RBT, and MWF were visually represented by plotting densities observed at each transect on maps of the survey area using GIS software.

RESULTS

WESTSLOPE CUTTHROAT TROUT

Westslope Cutthroat Trout were observed in 75% of the historic main-stem transects, with the highest linear densities occurring in the two upstream survey sections (Table 10). Mean linear density of WCT for all main-stem Lochsa River transects increased steadily from 1975 to 1981 and then dropped off in 2017 and 2018 (Table 10 and Figure 27). No significant trend (90% error bounds around estimates of r_{intr} overlapped zero) in mean linear density was detected for the overall (all transects) main-stem Lochsa River (Table 11).

Westslope Cutthroat Trout were observed in 82% of modern transects (since 2013), with the highest areal densities occurring in Colt Killed Creek (Table 12 and Figure 28). In 2018, mean areal density of WCT across all transects was similar to what was observed in 2017 (Figure 29). The largest change in areal density from 2017 to 2018 occurred in the WGB to Lake Creek and

Colt Killed Creek survey sections, with both sections experiencing at least four-fold increases in areal density in 2018 relative to 2013 surveys (Figure 29). In 2018, the mean areal density of WCT > 305 mm was four times higher than in 2017, but only one-third of the densities observed in 2013 (Figure 30). The proportion of WCT observed in 2018 that were > 305 mm TL for all transects in the Lochsa River drainage was 62%, the highest observed in the modern surveys (Table 13). The largest increase in areal density of WCT > 305 mm was observed in Crooked Fork Creek (Table 13 and Figure 30).

RAINBOW TROUT

Rainbow Trout had the lowest linear density of any study species observed in the main-stem Lochsa River (Table 10). They were observed in 68% of transects, with linear density the lowest in the Mouth to WGB survey section. Mean linear density increased in all three survey sections, with the largest increase occurring in the Lake Cr. - Crooked Fork Cr. survey section (Table 10). In 2018, mean linear density of RBT for all main-stem Lochsa River transects was the highest since 1981, increasing over 100-fold from 2017 (Table 10 and Figure 27). However, there was a statistically significant declining trend ($r_{intr} < 0$) in linear density of RBT from 1975 to 2018 (Table 11).

Mean areal density of RBT in the Lochsa River drainage was the lowest of any species (Table 12). They were observed in 71% of transects, with the highest areal densities occurring in Colt Killed Creek (Table 12 and Figure 31). In 2018, mean areal density of RBT for all transects was 40% of what was observed in 2017 (Figure 32). This decline in mean areal density was primarily the result of the decline observed in Colt Killed Creek, as areal densities increased in all main-stem survey sections (Figure 32). Rainbow Trout > 305 mm in TL accounted for 14% of those observed in the Lochsa River drainage in 2018, higher than both 2013 (3%) and 2017 (0%).

MOUNTAIN WHITEFISH

Mountain Whitefish were observed in 92% of transects that were snorkeled in 2018, with the highest areal densities occurring in Crooked Fork Creek (Table 12 and Figure 33). In 2018, mean areal density of MWF in all transects was the same as 2017, but was only about 25% of that observed in 2013 (Figure 34). Sixty-two percent of the MWF observed in 2018 were > 305 mm which was higher than observed in 2013 (50%) or 2017 (43%). Mean areal density of MWF > 305 mm was 50% higher in 2018 than 2017, but both years were < 35% of 2013 (Figure 35).

SMALLMOUTH BASS

One Smallmouth Bass (350 mm) was observed in the most downstream transect (LR01). This was the first time a SMB has been observed in the main-stem Lochsa River.

DISCUSSION

WESTSLOPE CUTTHROAT TROUT

No statistically significant trend in WCT linear density from 1975 to 2018 was detected in the main-stem Lochsa River. In some cases, the lack of a significant trend in linear density would suggest the population is stable. Despite the lacking statistical significant trend, snorkel survey data suggest the Lochsa WCT population may not be stable, as it increased dramatically from 1977 to 1981 and then declined in recent years (2017 and 2018). The increase in densities from 1977 to 1981 has been attributed to the implementation of catch-and-release regulations (Lindland 1982). In contrast, increasing trends in WCT linear density has been observed in the Selway, North Fork Clearwater (NFCR), North Fork Coeur d'Alene (CDAR), and St. Joe (SJR) rivers (Hand et al. 2020; Ryan et al. 2020; Hand et al. 2021). We must caution that our data series for the main-stem Lochsa River contains a 30-year gap (1982 - 2012) and therefore may not properly represent long-term trends.

The recent decline in WCT density in the Lochsa River is concerning. Snorkel surveys in the SJR and CDAR systems documented a short-term decline in WCT density after 2015, and indicated the low flow and higher temperatures observed in 2015 due to drought may have been responsible (Ryan et al. 2020; Camacho et al. 2021). Similar conditions occurred in the Lochsa River basin in 2015 and could explain for this recent decline. Because declines in WCT density were observed in multiple rivers in northern Idaho with different seasons and limits, it suggests the decline in density was driven by environmental conditions that either increased natural mortality or resulted in fewer fish being observed (fish moved to areas not snorkeled or were more difficult to see; Sloat et al. 2005; Copeland and Meyer 2011). The Lochsa River discharge at the time of sampling in 2013 (17.0 m^3/s) was < 50% of discharge in 2017 and 2018 (39.5 - 42.5 m^3/s; USGS 2021). This was primarily due to the timing of our surveys (early August in 2013 vs. mid-July in 2017 - 2018). The higher flows in 2017 and 2018 likely impacted detectability and indicates the need to conduct surveys at similar times of year and discharge. Additionally, discharge has been shown to have a positive correlation with trout survival on a 3 - 4 year time lag (Copeland and Meyers 2013). Mean annual discharge for the Lochsa River was below the 30-year average for three of the five years preceding 2018, including flow of approximately 50% of average in 2015. Thus, low mean annual discharge may have had a negative impact on survival in the years prior to sampling in 2017 and 2018. Changes in temperature can also have an impact on fish distribution and survival. Warmer summer temperatures have been found to result in fish migrating out of larger rivers and streams (e.g. most areas we surveyed) into colder streams and/or increase mortality where access to cooler waters is not available (Hunt 1992; Jager et al. 1999; Copeland and Meyer 2011; Kennedy and Meyer 2015). Severe winter conditions are also known to negatively impact salmonid populations in smaller streams by impacting redds and reducing survival of overwintering juvenile fish. Mean monthly summer air temperatures in north-central Idaho have been above normal every year except one since 1996, while they were > 2 °C below normal during the winter of 2016 - 2017 (NOAA 2021). These changes may have impacted fish populations in the Lochsa River through increased mortality and/or by moving fish out of our survey areas. The negative impacts of severe winter can include redd scour, displacement of fry, and reduced overwinter survival (Jager et al. 1999; Carline and McCollugh 2003; Zorn and Nuhfer 2007). Despite the recent decline, linear densities observed in 2017 and 2018 were more than double what was observed in the 1970s. Due to the importance of this fishery, the lower densities of WCT in recent surveys warrants continued monitoring. Additional surveys will allow for a more thorough evaluation of population trends.

Mean areal density of all sizes of WCT were lowest in the most downstream survey sections from the Mouth to WGB. Inherently, one may assume this is because WCT can be harvested downstream of WGB and not upstream of it; however, this is likely a response to warmer water temperatures in this survey section. Warmer water temperatures at lower elevations in many northern Idaho rivers have commonly been attributed to migrations of WCT into upstream reaches where cooler temperatures occur (Hunt 1982; Sloat et al. 2005; Copeland and Meyer

2011). In addition, the highest densities in our recent surveys were observed in Colt Killed Creek, which has some of the cooler water temperatures but harvest is allowed.

The areal density of WCT > 305 mm was higher than in 2017, but remained lower than was observed in 2013. The areal density (0.09/100 m^2) was lower than has been observed recently in the Selway River (0.28/100 m^2), North Fork Clearwater River (0.28/100 m^2), North Fork Coeur d'Alene River (0.24/100 m^2) and St. Joe River (0.40/100 m^2; Hand et al. 2020; Ryan et al. 2020). However, WCT > 305 mm comprised 62% of the individuals observed in the Lochsa River drainage. This was higher than observed in previous years, and twice as high as the proportion observed in the Selway River (25%) in 2018, and the North Fork Coeur d'Alene River (32%) and St. Joe River (28%) in 2017 (Hand et al. 2020; Ryan et al. 2020). The high proportion of larger fish observed in the Lochsa River may be due to a combination of low levels of harvest and poor recruitment resulting in few smaller fish. This warrants further attention as we conduct additional surveys of the Lochsa River.

RAINBOW TROUT

We have observed increasing RBT densities in historic surveys, but a substantial decline in recent surveys. This has led to a statistically significant declining trend in mean linear density of RBT in the main-stem Lochsa River from 1975 to 2018. Declining trends in RBT linear density has also been observed in the NFCR, Selway River, CDAR, and SJR (Hand et al. 2020; Ryan et al. 2020; Hand et al. 2021). We must caution that our data series for the main-stem Lochsa River contains a 30-year gap (1982 - 2012) and therefore may not properly represent long-term trends in density. The ~97% decline in linear density observed in 2013 and 2017 compared to previous surveys was alarming. Although linear density increased in 2018, it was still six times lower than observed in 1981. Declining trends in RBT have been primarily been attributed to hybridization, increasing temperatures from climate change, and habitat degradation (Zoellick 2005; Meyer et al. 2014; Muhlfeld et al. 2015). While these may be contributing factors in the Lochsa River, a major factor is likely the change in hatchery stocking practices during this time period. From 1968 to 1990, IDFG annually stocked ~11,000 RBT > 150 mm into the Lochsa River. No RBT have been stocked since, which would contribute to the decline in densities observed in surveys conducted since 1990. Additionally, recent declines in steelhead smolt out-migration and adult returns have been observed throughout the Clearwater River basin, and specifically within the Lochsa River basin (Dobos et al. 2020; Feeken et al. 2020). We have previously attributed declines in RBT within the North Fork Clearwater River drainage to the loss of steelhead from the construction of Dworshak dam (Pettit 1976; Hand et al. 2016). However, wild steelhead adult returns were higher in the 1970s than 2010 - 2016. Thus, while lower returns are likely impacting current densities, it would not explain the decline observed in modern surveys. Additional surveys will allow for a more thorough analysis of trends in RBT density in the future.

Few RBT > 305 mm have been observed in the Lochsa River basin in surveys since 2013. This size distribution is similar to other rivers where steelhead occur (*see* Selway River and South Fork Clearwater River chapters in this report). The size of RBT observed in our recent surveys is comparable to the size of juvenile steelhead out-migrating from Fish Creek (major tributary of the Lochsa River) from 1995 to 2017, which had a maximum TL of ~220 mm (Dobos et al. 2020). Thus, the lack of larger RBT in the Lochsa River drainage is to be expected, as many of those observed in surveys are likely juvenile and residualized steelhead. With so few large fish, the RBT population in the Lochsa River basin is mostly unavailable to the harvest fishery and would therefore be only minimally affected by angling.

Differences in RBT density between survey sections is most likely influenced by water temperature, as harvest is not a primary contributing factor. The highest densities were observed in Colt Killed and Crooked Fork creeks, the upstream-most sections, which are cooler than downstream sections. Steelhead have been found to move out of tributaries in the fall, then move back into these locations in the summer (Bjornn 1971; Dobos et al. 2020; Knoth et al. 2020). This migration pattern would account for higher densities observed in the upper survey sections. The large changes in densities between Colt Killed and Crooked Fork creeks from 2013 to 2018 was likely due to annual variation in distribution and low number of survey transects (five each). Densities estimated during NPM snorkel surveys in these years were more stable (ranged from 0.8 to 1.1/100 m^2; Stiefel et al. 2014; Putnam et al. 2018; Roth et al. 2019). The NPM surveys include more transects and cover larger areas of each drainage. Thus, they are more likely to account for annual variation in distribution.

<u>MOUNTAIN WHITEFISH</u>

Mean areal density of MWF (all sizes) was the same in 2017 and 2018, but was 25% of that observed in 2013. Mean areal density of MWF > 305 mm was higher in 2018 than in 2017, but both years were < 35% of 2013. Areal density in the Lochsa River (0.18/100 m^2) was lower than observed in the South Fork Clearwater River (0.49), and the Selway River. However, there still appears to be a declining trend in MWF density. While our data for MWF in the Lochsa River is short-term, declines in densities have been observed in the main-stem of other northern Idaho rivers, including the Selway River and South Fork Clearwater River (*See* Selway River and SFCR chapters in this report). The only other large data set we have on MWF in the Lochsa River basin is from an intensive snorkel survey conducted in 2003, where the mean areal density (2.2/100 m^2) was higher than surveys conducted from 2013 to 2018 (Hand et al. 2016). While not directly comparable (due to utilizing different snorkel transects as other surveys), it does suggest mean density of MWF may be declining in the main-stem Lochsa River.

Long-term declines in MWF have been documented in other populations across the southern portion of their range as well, including the Big Lost River and Kootenai River, Idaho, the Yampa River, Colorado, and the Madison River, Montana (Paragamian 2002; IDFG 2007; Boyer 2016). Although these surveys occurred during a different time period, and thus are not directly comparable to our data, they suggest that habitat alteration, irrigation, nonnative fish interactions, disease, and harvest are likely contributing to declines in MWF populations (IDFG 2007; Brinkman et al. 2013; Boyer 2016). While the direct cause of these declines has not been identified, these declines have been linked to occurrences of low flows and higher water temperatures (Brinkman et al. 2013). Increased water temperatures could affect MWF populations by moving more fish out of the main-stem and larger tributaries (where we surveyed) and/or through increased mortality (Jager et al. 1999; Copeland and Meyer 2011; Kennedy and Meyer 2015). Mean monthly summer air temperatures in north-central Idaho have been above normal every year except one since 1996 (NOAA 2021). As such, increased water temperatures could impact MWF movement, survival, and recruitment before some other species. Disease is another issue to consider. Severe outbreaks of Proliferative Kidney Disease (PKD) have been observed in Montana, and the disease is known to be present in Idaho (Phillips 2016; Hutchins et al. 2021). While no major fish kills have been directly observed, minor die-offs have been observed in in Idaho rivers during summer months. Thus, PKD could also impact populations through lower level mortality.

As snorkel surveys continue into the future, we will be able to better evaluate the status of MWF in the Lochsa River drainage.

SMALLMOUTH BASS

Smallmouth Bass were observed during our sampling in the Lochsa River for the first time. The one SMB observed was 350 mm, indicating that there is not successful recruitment occurring in the Lochsa River at this time. In previous years, SMB have also been observed in the lower reaches of other Clearwater River tributaries, including the North Fork Clearwater and South Fork Clearwater rivers (Hand et al. 2020; *see* South Fork Clearwater River section). The lower North Fork Clearwater River is at a higher elevation (503 m) than the lower Lochsa River (448 m). Additionally, SMB have been found in numerous other locations at elevations higher than the Lochsa River, such as the Flathead River (790 m), Montana, the Salmon River (911 m) near Idaho Falls, Idaho, and the Owyhee River (1,030 m) upstream of Owyhee Reservoir (Rubenson and Olden 2017). This indicates that SMB could utilize the lower reaches of the Lochsa River. Potential increases in SMB distribution is of concern, as these non-native fish can be a substantial predator of juvenile anadromous and resident salmonids and could impact native fish populations (Tabor et al. 1993; Naughton et al. 2004; Tiffan et al.2020). Smallmouth Bass colonization of salmonid spawning and rearing habitat has been documented throughout the Columbia River Basin (Lawrence et al. 2014; Rubenson and Olden 2017). At this time, few SMB appear to occur in the Lochsa River system. However, future surveys should continue to evaluate the distribution of these non-native fish, as they may experience a climate-mediated spread throughout the upper Clearwater River system (Rahel and Olden 2008).

MANAGEMENT RECOMMENDATIONS

1. Continue to evaluate trends in density and their size structure of game fishes in the Lochsa River drainage on a two year on, two year off basis, and assess whether season and limits and/or environmental factor play a role in the trends that are being observed.

Table 10. Comparisons of Westslope Cutthroat Trout (WCT) and Rainbow Trout (RBT) linear densities (fish/100 m) determined through snorkel surveys conducted in the Lochsa River, Idaho, from 1975 to 2018.

WCT

	Survey section			
Year	Mouth of Locsha to Fish Creek	Fish Creek to Lake Creek	Lake Creek to Crooked Fork Creek	Overall
1975	0.00	0.00	0.00	0.00
1976	0.16	0.07	0.00	0.08
1977	0.08	0.08	1.15	0.19
1978	0.11	1.17	2.20	0.82
1979	0.13	0.41	3.04	0.68
1980	0.33	5.00	8.00	3.73
1981	0.50	3.75	6.67	4.00
2013	0.02	0.46	0.37	0.20
2017	0.64	0.96	3.32	1.40
2018	0.51	2.48	2.74	1.63

RBT

	Survey section			
Year	Mouth of Locsha to Fish Creek	Fish Creek to Lake Creek	Lake Creek to Crooked Fork Creek	Overall
1975	0.37	2.50	1.14	1.34
1976	0.04	2.70	3.72	2.15
1977	4.23	7.60	0.00	2.60
1978	2.56	4.46	0.00	2.89
1979	4.38	1.18	0.68	2.61
1980	0.66	11.00	0.00	4.67
1981	11.00	10.25	0.00	7.00
2013	0.08	0.04	<0.01	0.02
2017	<0.01	0.01	0.01	0.01
2018	0.26	1.79	2.17	1.18

Table 11. Intrinsic rate of population change (r_{intr}) for Westslope Cutthroat Trout (WCT) and Rainbow Trout (RBT) in the Lochsa River basin, Idaho, from 1975 to 2018. Significance was set at $\alpha = 0.10$.

	r_{intr}	90% CI	
Species	estimate	lower	upper
WCT	0.073	-0.030	0.176
RBT	-0.096	-0.146	-0.045

Table 12. Number and density of fish observed while snorkeling transects in the Lochsa River drainage, Idaho, during 2018.

Survey section	Transect name	Area (m^2)	Temp (°C)	Visability (m)	Number of fish			Density (fish/100 m^2)		
					WCT	RBT	MWF	WCT	RBT	MWF
Mouth to Wilderness	LR01	13,608	20.0	2.4	0	0	0	0.00	0.00	0.00
Gateway Bridge	LR02	14,414	20.0	2.6	0	0	3	0.00	0.00	0.02
	LR03	26,633	20.0	1.9	2	0	5	0.01	0.00	0.02
	LR04	17,168	20.0	3.3	1	0	29	0.01	0.00	0.17
	LR05	14,310	20.0	3.1	1	0	5	0.01	0.00	0.03
	LR06	15,660	20.0	3.4	0	0	3	0.00	0.00	0.02
	LR07	13,230	20.0	3.2	0	0	5	0.00	0.00	0.04
	LR08	9,860	18.0	2.9	0	3	9	0.00	0.28	0.85
	LR09	2,146	18.0	2.9	5	4	5	0.23	0.19	0.23
	LR10	1,056	18.0	2.9	4	1	14	0.04	0.01	0.14
	LR11	4,810	19.0	3.0	0	2	9	0.00	0.04	0.19
	LR12	5,805	20.0	3.4	2	1	3	0.03	0.02	0.05
	LR13	6,150	20.0	3.4	1	2	4	0.02	0.03	0.07
Wilderness Gateway	LR14	4,000	20.0	2.4	0	1	0	0.00	0.03	0.00
Bridge to Lake Creek	LR15	4,326	20.0	2.4	4	4	8	0.09	0.09	0.18
	LR16	2,408	20.0	1.9	3	1	4	0.12	0.04	0.17
	LR17	2,400	18.0	2.9	10	12	2	0.42	0.50	0.08
	LR18	5,100	16.5	2.7	8	4	5	0.16	0.08	0.10
	LR19	17,680	17.0	2.9	12	6	11	0.07	0.03	0.06
	LR20	10,249	18.0	1.8	11	5	34	0.11	0.05	0.33
	LR21	9,128	17.0	3.2	14	3	28	0.15	0.03	0.31
Lake Creek to	LR22	7,412	15.0	2.8	2	4	0	0.03	0.05	0.00
Crooked Fork Creek	LR23	8,134	18.0	3.0	13	8	13	0.16	0.10	0.16
	LR24	4,644	18.0	---	1	0	16	0.02	0.00	0.34
	LR25	4,818	18.5	3.7	7	5	11	0.15	0.10	0.23
	LR26	5,100	19.0	3.1	7	0	12	0.14	0.00	0.24
	LR27	5,400	19.0	3.9	17	21	19	0.31	0.39	0.35
	LR28	10,500	14.0	2.5	4	2	6	0.04	0.02	0.06
Colt Killed Creek	CKC01	8,820	18.0	3.8	1	1	5	0.01	0.10	0.05
	CKC02	4,000	18.0	3.9	2	4	5	0.23	0.00	0.53
	CKC03	4,080	17.0	3.3	4	10	7	0.05	0.17	0.02
	CKC04	4,064	16.0	2.8	5	5	9	0.07	0.05	0.05
	CKC05	4,050	15.0	3.8	9	31	21	0.10	0.00	0.27
Crooked Fork Creek	CFC01	2,075	12.0	3.8	1	2	4	0.05	0.01	0.24
	CFC02	2,070	13.0	3.9	9	0	21	0.10	0.10	0.24
	CFC03	2,415	13.0	3.3	2	4	1	0.17	0.25	0.29
	CFC04	1,820	17.0	2.8	3	1	2	0.29	0.12	0.51
	CFC05	1,300	18.0	3.8	4	0	11	0.69	0.77	1.62
	Mean				4.4	3.9	9.2	0.11	0.10	0.22
	90% CI				0.0	0.2	0.1	0.04	0.04	0.08

WCT - Westslope Cutthroat Trout; RBT - Rainbow Trout; MWF - Mountain Whitefish.

Table 13. Percent of Westslope Cutthroat Trout > 305 mm observed in snorkel surveys conducted in the main-stem Lochsa River, Idaho, from 2013 to 2018.

Survey section	2013	2017	2018
Mouth - Wilderness Gateway Bridge	59	47	70
Wilderness Gateway Bridge - Lake Cr.	45	19	67
Lake Cr. - Crooked Fork Cr.	50	60	49
Crooked Fork Cr.	59	7	63
Colt Killed Cr.	52	40	71
Main-stem mean	49	47	60
Drainage mean	50	43	62

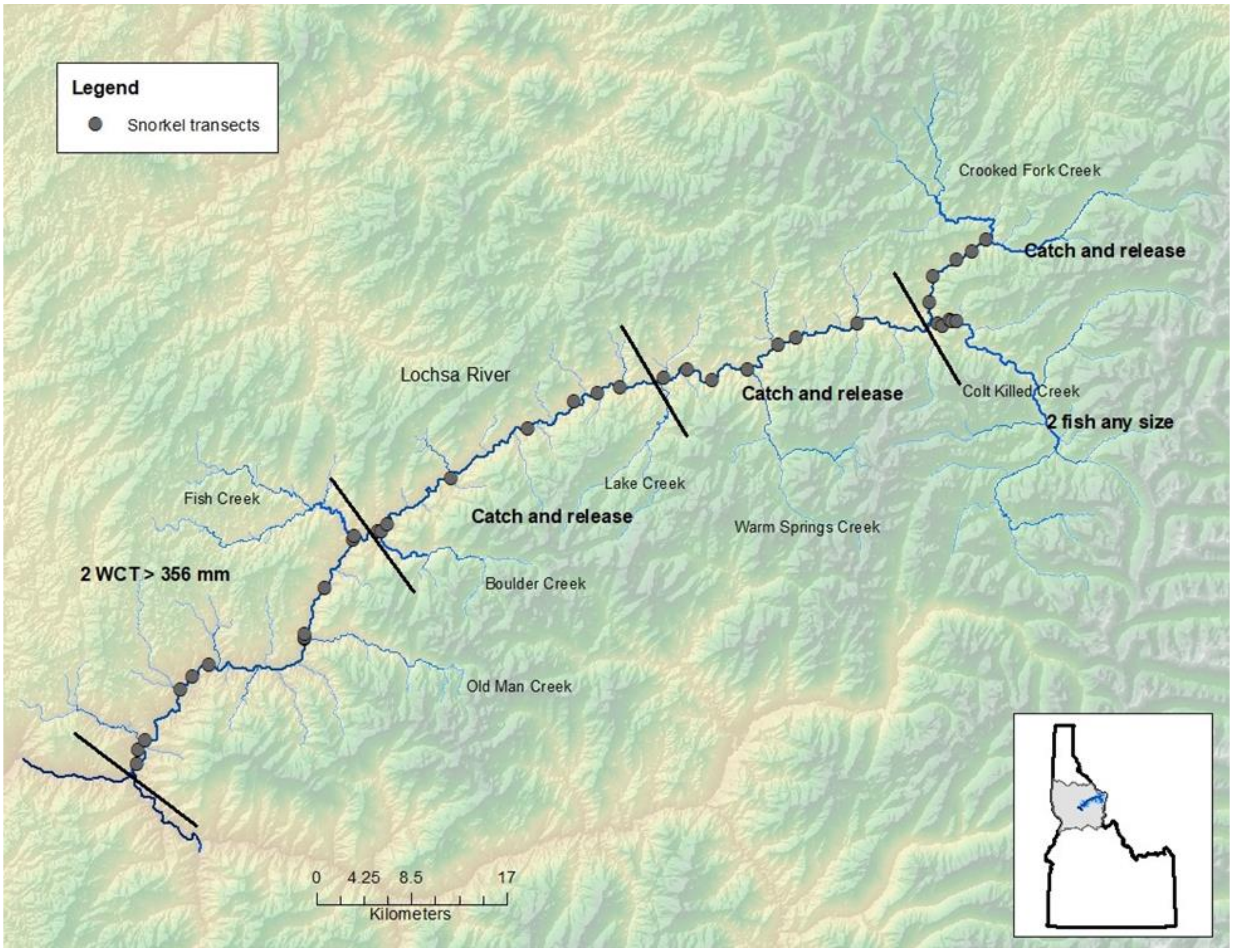

Figure 26. Map showing locations of snorkel transects surveyed in the Lochsa River drainage, Idaho, in 2018. Black bars delineate survey sections.

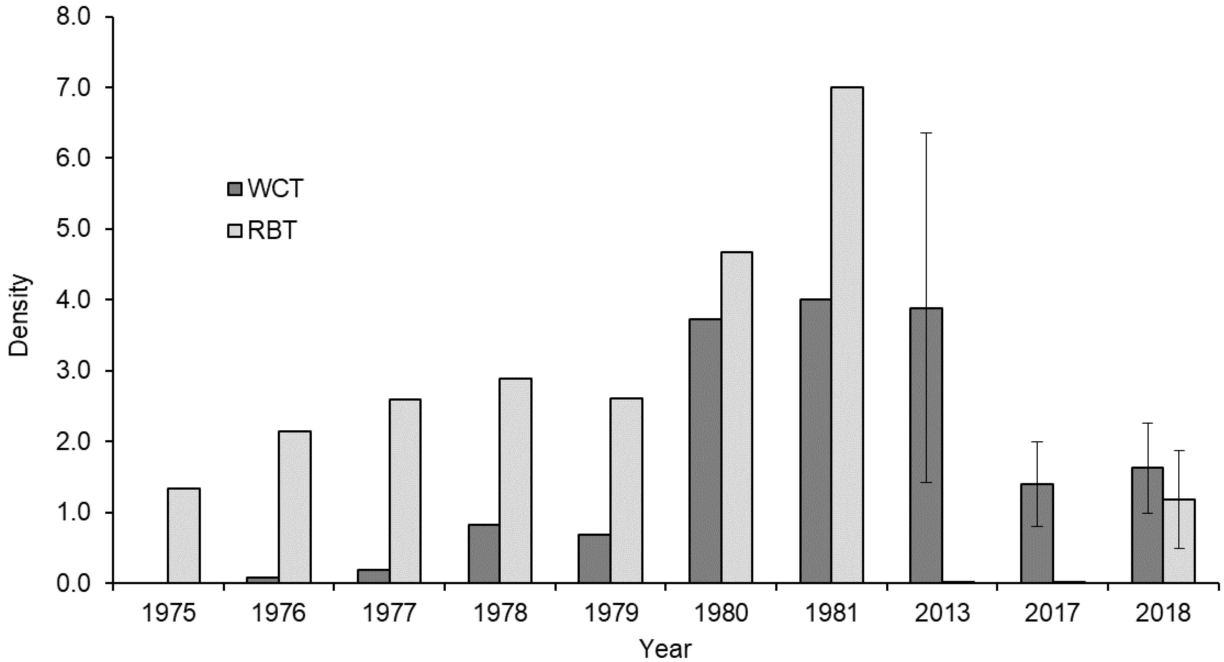

Figure 27. Comparisons of mean linear density (fish/100 m) of Westslope Cutthroat Trout (WCT) and Rainbow Trout (RBT) observed during snorkel surveys of the main-stem Lochsa River, Idaho, from 1975 to 2018. Error bars represent 90% confidence intervals, and could only be calculated for 2013 - 2018.

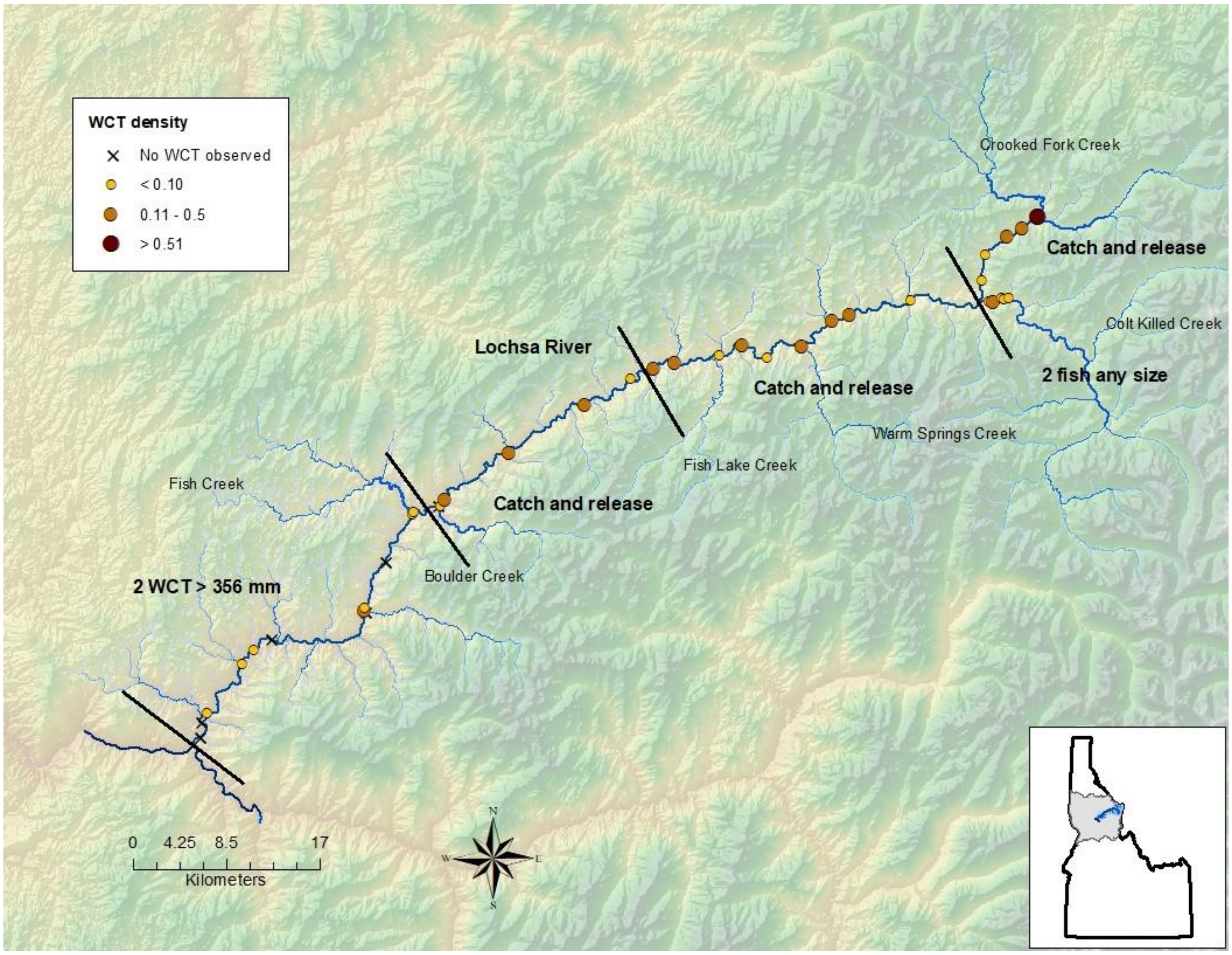

Figure 28. Densities (fish/100 m^2) of Westslope Cutthroat Trout (WCT) observed at each snorkel transect surveyed in the Lochsa River basin, Idaho, in 2018. Black bars delineate survey sections.

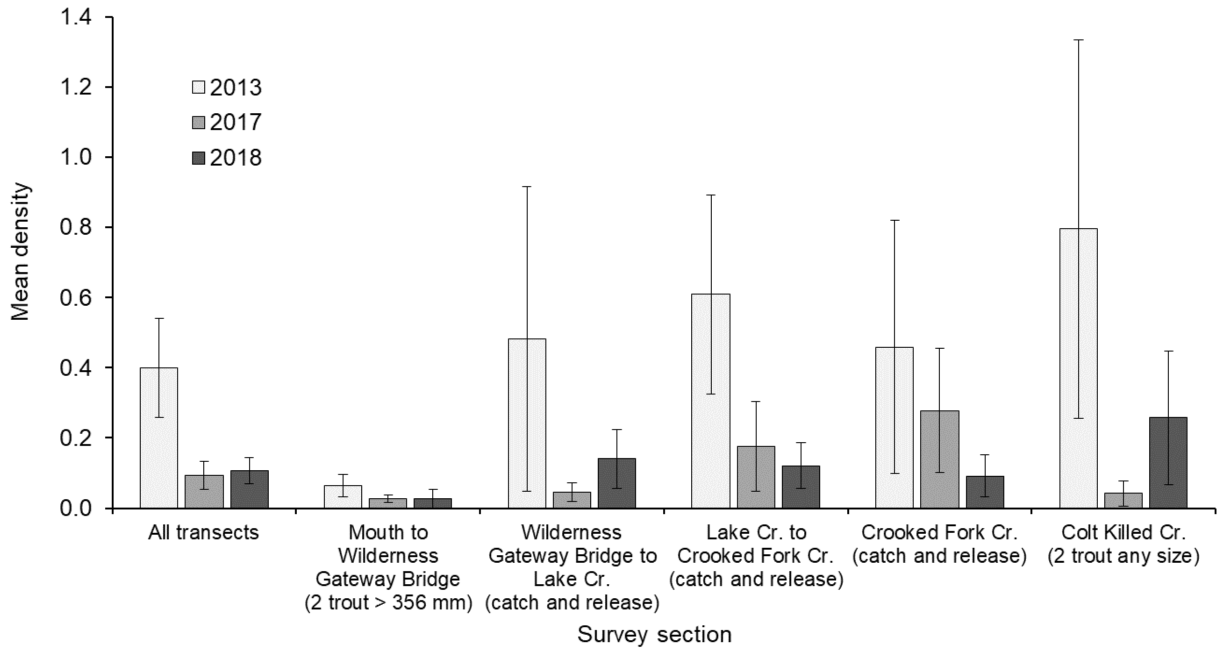

Figure 29. Comparisons of mean densities (fish/100 m^2) of Westslope Cutthroat Trout in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys from 2013 to 2018. Error bars represent 90% CIs.

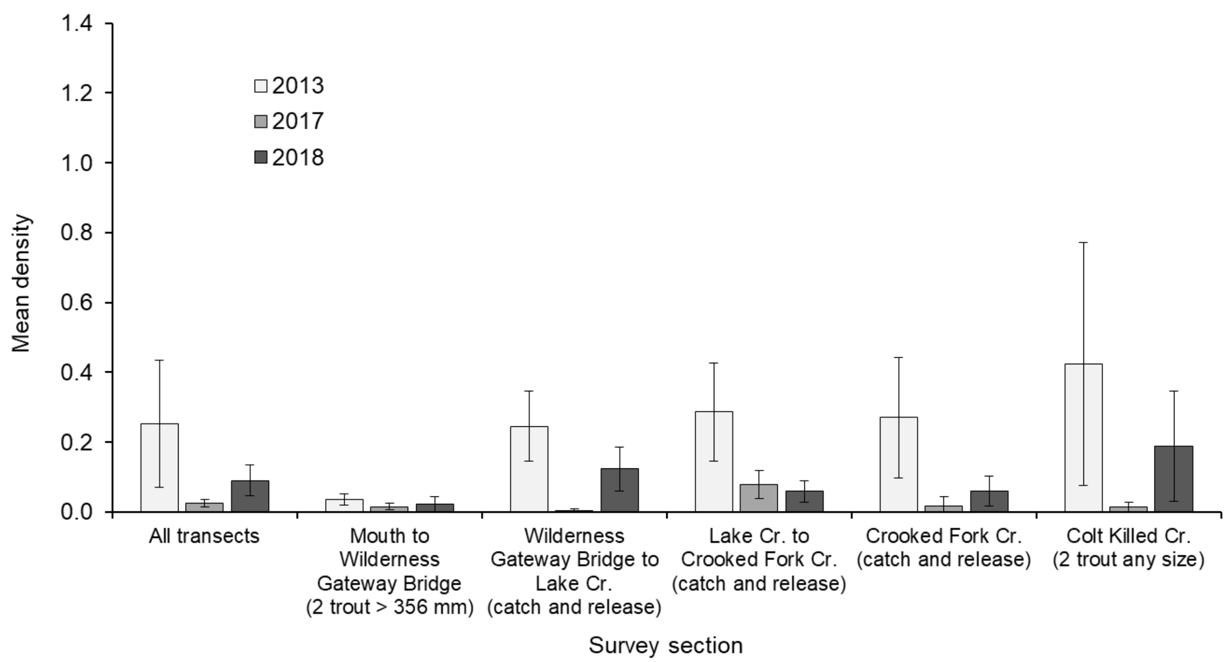

Figure 30. Comparisons of mean densities (fish/100 m^2) of Westslope Cutthroat Trout > 305 mm in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys from 2013 to 2018. Error bars represent 90% CIs.

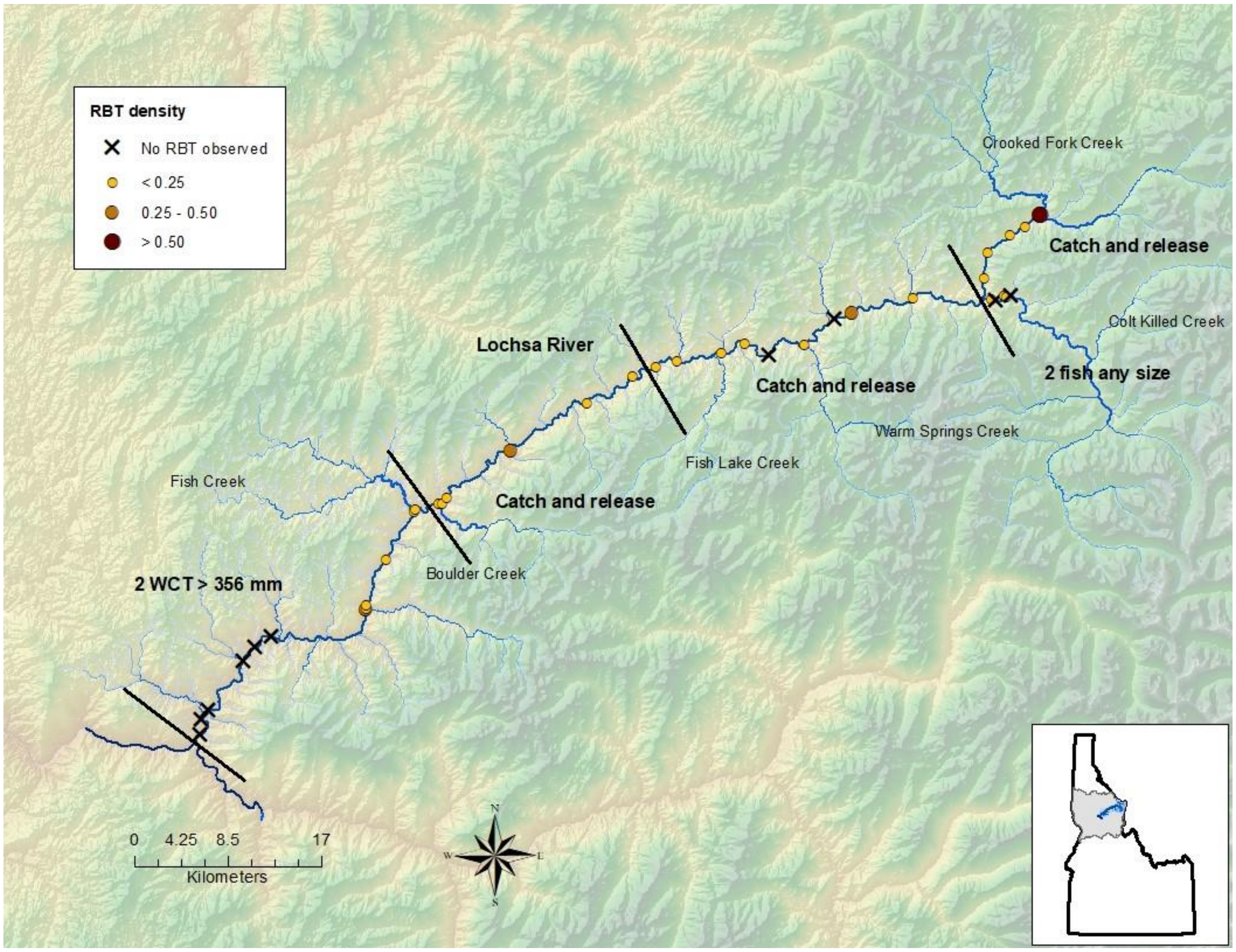

Figure 31. Densities (fish/100 m^2) of Rainbow Trout (RBT) observed at each snorkel transect surveyed in the Lochsa River basin, Idaho, in 2018. Black bars delineate survey sections.

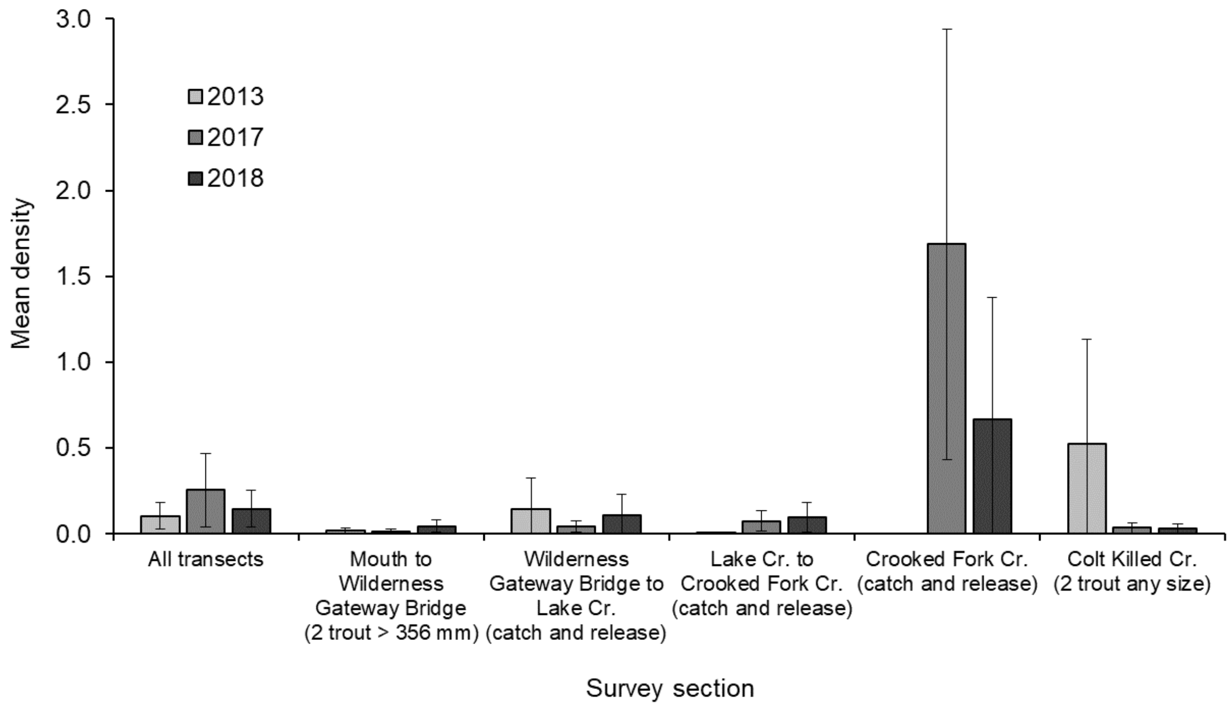

Figure 32. Comparisons of mean densities (fish/100 m^2) of Rainbow Trout in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys from 2013 to 2018. Error bars represent 90% CIs.

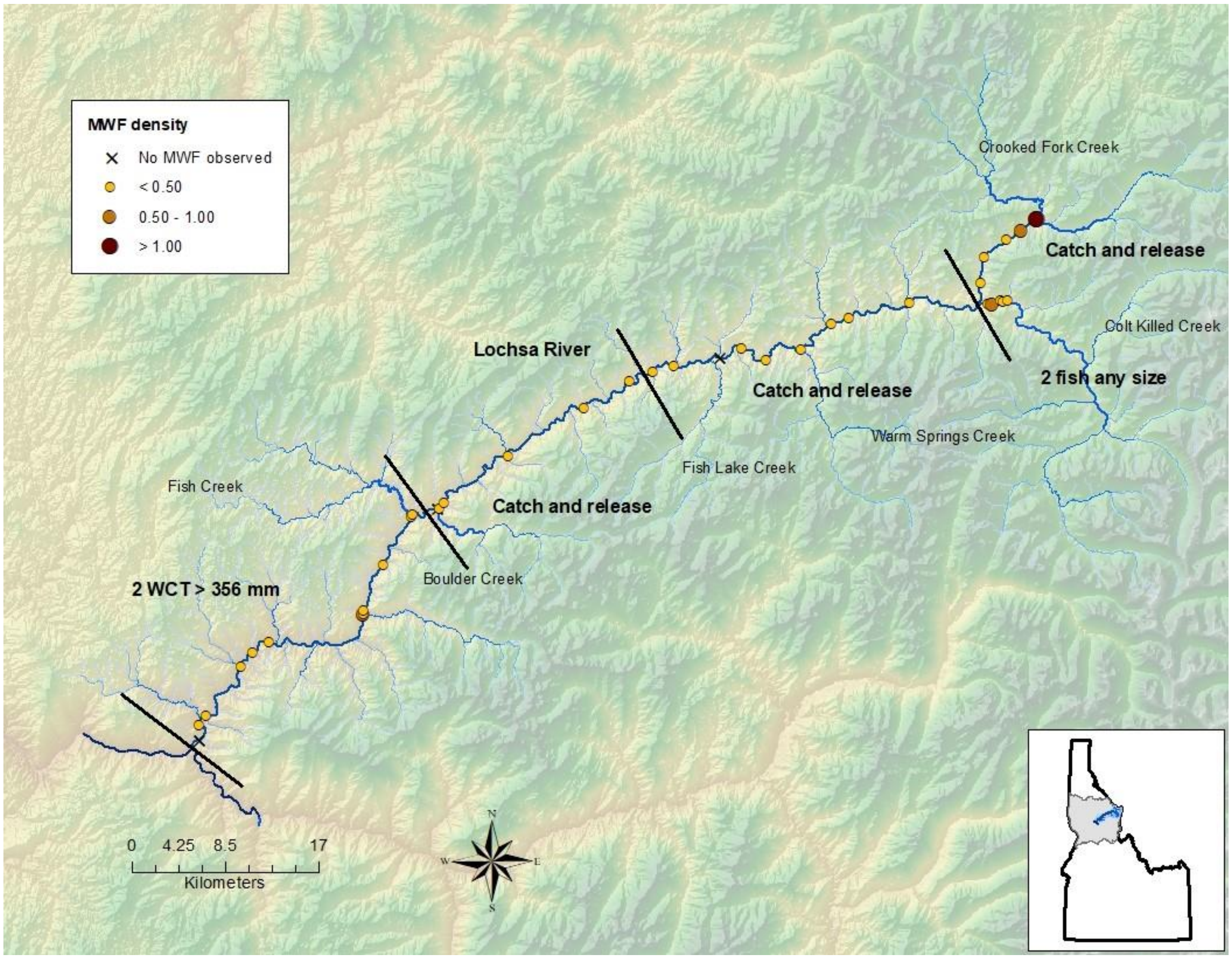

Figure 33. Densities (fish/100 m^2) of Mountain Whitefish (MWF) observed at each snorkel transect surveyed in the Lochsa River basin, Idaho, in 2018. Black bars delineate survey sections.

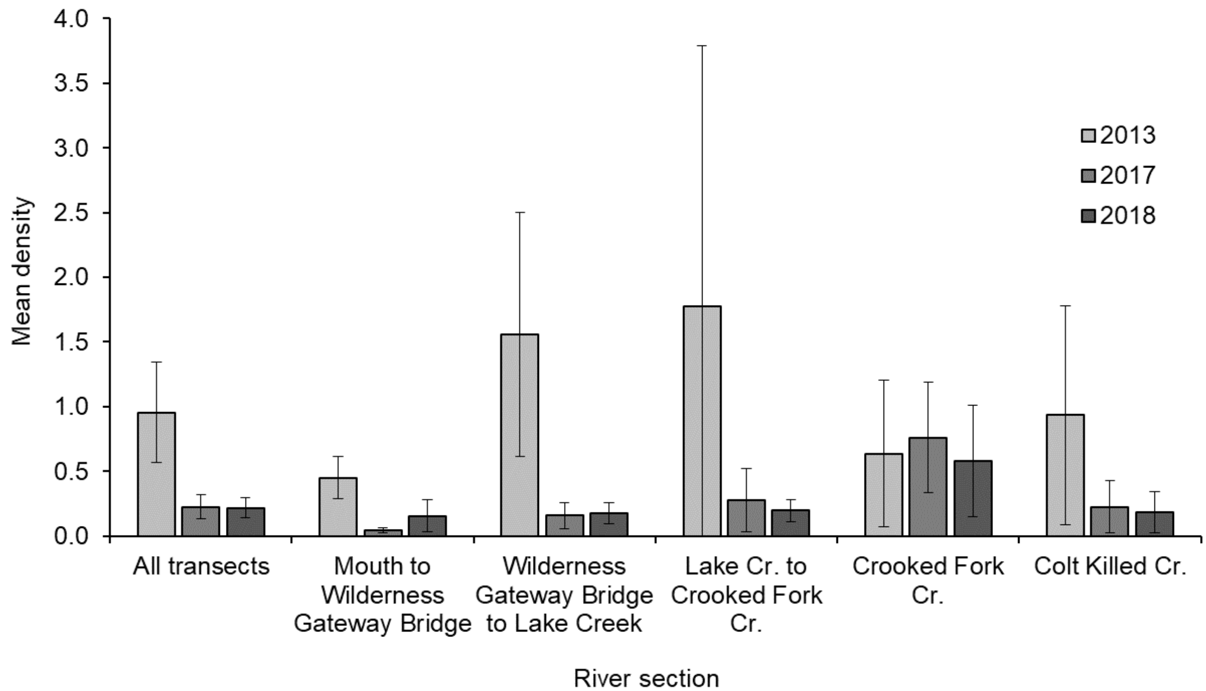

Figure 34. Comparisons of mean densities (fish/100 m^2) of Mountain Whitefish in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys from 2013 to 2018. Error bars represent 90% CIs.

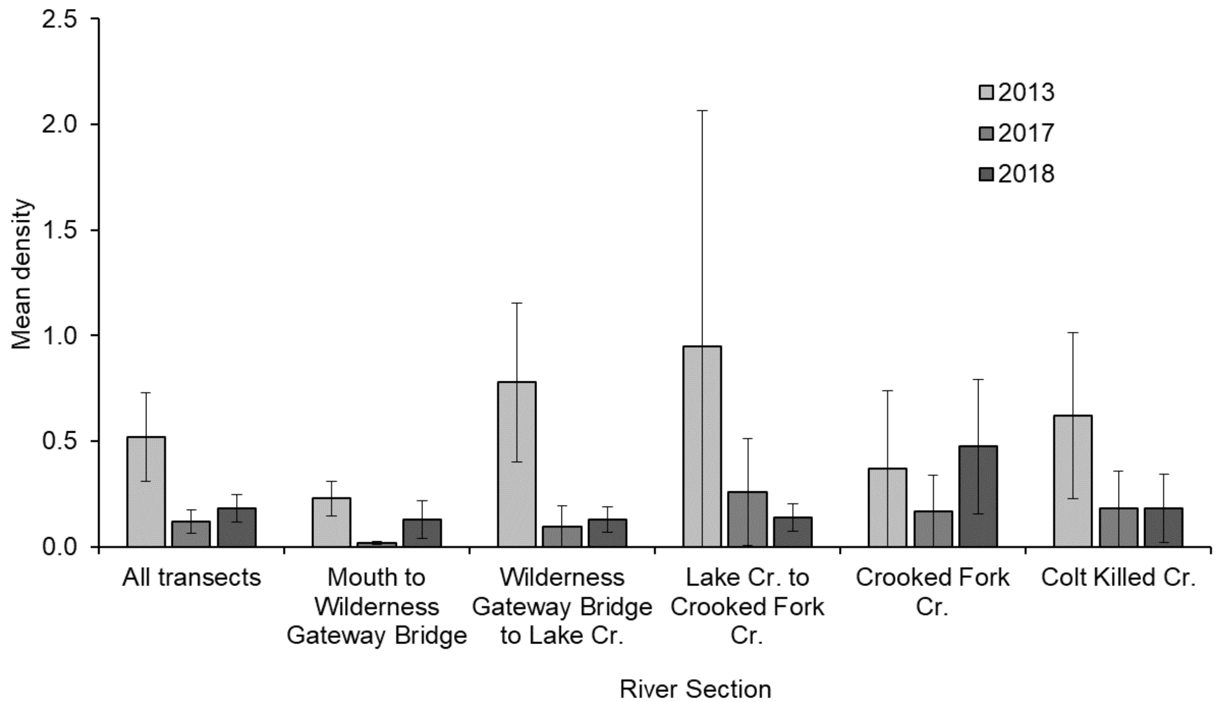

Figure 35. Comparisons of mean densities (fish/100 m^2) of Mountain Whitefish > 305 mm in survey sections of the Lochsa River drainage, Idaho, observed during snorkel surveys from 2013 to 2018. Error bars represent 90% CIs.

LITERATURE CTED

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EVALUATION OF FISH POPULATIONS IN THE SELWAY RIVER

ABSTRACT

Snorkel surveys were conducted in the Selway River drainage in 2018 to assess trends in Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT), Rainbow Trout *O. mykiss* (RBT), and Mountain Whitefish *Prosopium williamsoni* (MWF) density and size distribution. One-person surveys were conducted on the main-stem river, while General Parr Monitoring surveys were conducted on both the main-stem and tributaries. There was not a significant long-term trend in WCT density (all fish and just those > 305 mm) in the main-stem or tributaries of the Selway River. There was a significantly declining trend in RBT linear density in 1-person transects since 1973, and in General Parr Monitoring transects since 1992. Since 1988, there has been a significant declining trend in MWF areal density in tributaries of the Selway River, but a significant increasing trend in the main-stem river. There was no correlation between snorkel survey types for all main-stem transects, but there was some correlation when only comparing transects between Bear and White Cap creeks. While hook-and-line catch rates of WCT have remained stable, there were increasing trends in both mean total length and proportion of fish > 305 mm caught. Overall, our data indicates that the WCT population in the Selway River is stable or increasing in density and in proportion of fish > 305 mm, and is therefore meeting our management objectives. The decline in RBT density primarily occurred prior to 1990, when ~5,000 RBT were stocked annually in the Selway River. However, since RBT in the Selway River are likely primarily juvenile and residualized steelhead, the trends in adult steelhead returns to Idaho are likely having an impact as well. The primary drivers of MWF population trends in the Selway River are likely environmental factors affecting movement and recruitment. In spite of the long-term increasing trend in the main-stem river, the recent declining trend in MWF densities across northern Idaho rivers and other parts of their historic range warrants a more thorough evaluation. The lack of correlation between survey types is partially due to the distribution and number of each type of survey. We recommend maintaining consistent sampling practices and ensuring snorkelers are well trained. We also recommend evaluating observer bias in 1-person surveys by snorkeling these sites multiple times with different snorkelers. Catch rates of WCT during the hook-and-line survey and the percentage of WCT > 305 mm are stable or increasing, indicating we are meeting our management goal of providing a high quality fishery. However, the recent decline in the number of smaller fish caught is concerning, as it may be an indicator of reduced recruitment.

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INTRODUCTION

Westslope Cutthroat Trout *Oncorhynchus clarkii lewisii* (WCT) are distributed throughout the Selway River drainage, occupying both the main river and tributaries. Both resident and fluvial life history forms are present. Fish populations in the Selway River have been regularly evaluated through snorkel surveys from White Cap Creek downstream to Selway Falls since 1973. Early studies of WCT in other northern Idaho rivers such as the St. Joe River, Kelly Cr., and the Lochsa River, concluded that the low WCT densities were a result of overfishing (Mallet 1967; Dunn 1968; Rankel 1971; Lindland 1977a). Concerns over declining WCT populations prompted IDFG to implement catch-and-release regulations in the Selway River in 1976 (Lindland 1977b). Subsequent surveys showed that WCT abundance tripled over the four-year period after catch-and-release regulations were implemented. Similar trends in WCT abundance were also observed after catch-and-release regulations were implemented in the St. Joe River, Kelly Creek, and Lochsa River (Lindland 1977a).

After peaking in 1986, WCT counts have fluctuated, likely in response to drought, temperature extremes, flooding, and observer variability. Similar long-term fluctuations in WCT densities have also been observed in other Idaho rivers (Flinders et al. 2013; Ryan et al. 2020). As the majority of this watershed is afforded protected status through wilderness or roadless designations, land management and human development have little influence on WCT abundance. Limiting factors for WCT are therefore closely tied to climatic regimes.

Due to limited vehicle access to this watershed, fishing effort on the Selway River and its tributaries is relatively light. Currently, the fishery in the Selway watershed is managed under three different fishing rules. In all tributaries, a daily limit of two WCT is allowed. The regulation on the main-stem Selway River for WCT are catch-and-release except for downstream of Selway Falls where a daily limit of two WCT > 356 mm is allowed from Memorial Day weekend to November 30. Downstream of Selway Falls also receives the most recreation as it is accessible by road. However, WCT use is limited in this reach of the Selway River during much of the summer due to unsuitable water temperatures. For these reasons, impacts from fishing are believed to have minimal influence on this WCT population. Monitoring this WCT population is important as it provides insight to trends in abundance in a watershed with light fishing effort, limited harvest, and little influence from land management activities and can be used as a baseline when comparing to other riverine fisheries in Northern Idaho.

OBJECTIVES

1. Evaluate trends in density and size structure of WCT, Rainbow Trout *O. mykiss* (RBT) and Mountain Whitefish *Prosopium williamsoni* (MWF) in the Selway River.

2. Evaluate 1-person vs. General Parr Monitoring survey types to determine if trends in these data sets correlate with one another.

STUDY AREA

The Selway River flows ~163 km from its headwaters in the Bitterroot Mountain Range to its confluence with the Lochsa River where it forms the Middle Fork Clearwater River (Figure 36). The Selway River watershed encompasses an area of ~5,200 km^2. The majority of the watershed occurs at elevations > 1,200 m. Land ownership in the Selway River watershed is almost 100%

Federal and is managed by the U.S. Forest Service. About 95% of the watershed is afforded some level of protected status, primarily as wilderness (Selway Bitterroot and Frank Church River of No Return Wildernesses) or roadless areas. The Selway River has a road that parallels its path for the lower 30 km and about 15 km in the upper reaches.

METHODS

FIELD SAMPLING

Snorkel survey

Westslope Cutthroat Trout, RBT, and MWF populations in the Selway River basin were surveyed through snorkeling 50 transects from July 12 to 19, 2018 (Figure 36). Two types of transects were snorkeled. The first group of transects are part of the General Parr Monitoring (GPM) program, developed to estimate anadromous fish response to Bonneville Power Administration habitat improvement projects (Scully et al. 1990). These transects are located on both the main-stem river and tributaries, and use standard snorkeling methodologies outlined in Apperson et al. (2015). This technique entails using an appropriate number of snorkelers to cover the entire width of the river to allow for the calculation of fish densities. They were conducted downstream in the main-stem river, and upstream in tributaries. The second group of surveys were 1-person transects developed for monitoring trends in density and size distribution of resident fish such as WCT and MWF. These transects are located on the main-stem river, and utilize one person starting at the upstream end of the transect and snorkeling downstream through the thalweg. Locations (GPS coordinates) and photographs of each 1-person and GPM snorkel transect are provided in Appendices “L” and “M” of DuPont et al. (2011). Of the 50 transects surveyed in 2018, 26 were historic GPM transects and 24 were 1-person transects.

For both types of snorkel surveys, all fish observed were counted, and total length (TL) was estimated to the nearest inch for all game species. Other species (e.g. *Cottus* spp, *Catostomus* spp.) were categorized as > or < 305 mm. Transect length (m) and average width (m; based on five measurements) was measured using a Nikon ProStaff S laser rangefinder. Visibility (m) was estimated at each transect by holding a Keson 50-m, reel style, fiberglass measuring tape underwater. A snorkeler backed away from the reel until lettering was indistinguishable, then moved back towards the reel until the lettering was viewable again. The distance from snorkeler to the reel was recorded. Habitat type, date, time of day, water temperature, and weather conditions were also recorded for each transect. Juvenile steelhead and resident Rainbow Trout are indistinguishable, and are collectively referred to as “RBT”. This report focuses primarily on WCT, RBT, and MWF. Results and analysis of data collected on other species in 2018 can be found in Roth al. (2019).

Hook-and-line survey

Main-stem Selway River fish populations were surveyed by hook-and-line from July 12 to 19, 2018, while rafting from White Cap Creek to Race Creek (just upstream of Meadow Creek). Anglers utilized both fly and lure (spinners and spoons) techniques. Gear type, species, and TL (mm) were recorded for each fish captured. Anglers also noted any potential mortalities. Angler effort was recorded daily for each raft, but is estimated, as it is extremely difficult to calculate accurately due to the numerous interruptions that occur when rafting a technical river at low water, and the many stops required to conduct snorkel surveys.

DATA ANALYSIS

Snorkel survey

Snorkel survey data was organized and summarized by river sections: tributaries, and main stem sections from Race Cr. to Three-Links Cr., Three-Links Cr. to Moose Cr., Moose Cr. to Bear Cr., Bear Cr. to Running Cr., Running Cr to White Cap Cr., and above White Cap Cr. In this report, we refer to density in 1-person transects (fish/transect) as "linear density", and in GPM transects (fish/100 m^2) as "areal density". Mean density for each year was calculated as the mean of all transects surveyed in a given year. Analysis of 1-person transects was conducted on the available data as follows: all sizes of WCT, RBT, and MWF, and WCT > 305 mm (1973 - 2018); RBT and MWF > 305 mm (2002 - 2018). Analysis of GPM transects for WCT, RBT, and MWF was conducted on the available data as follows: main-stem (1992 - 2018); tributaries (1988 - 2018). The length of time series available for each data set varied due to differences in when data collection started, and how data was collected/summarized. We evaluated trends in WCT, RBT, and MWF linear density in 1-person transects and areal density in GPM transects for all observed fish and just those > 305 mm using least squares regression. We used survey year as the independent variable and $\log_e$ transformed density as dependent variables (Maxell 1999; Kennedy and Meyer 2015). The intrinsic rate of change in the population (r_{intr}) was determined by the slope of the regression line fit to these data. A 90% CI was calculated for r_{intr} to determine significance, where the trend is considered significant when $r_{intr} \neq 0$ and the error bounds do not include 0. We used a significance level of $\alpha = 0.10$. Distributions of WCT, RBT, and MWF were visually represented by plotting mean density for each transect on maps of the survey area using GIS software.

To evaluate if trends in the 1-person and GPM data sets correlated with one another, we plotted WCT annual mean linear density in 1-person surveys (x-axis) against the annual mean areal density of GPM surveys (y-axis), for surveys conducted from 1992 to 2018 in the main-stem Selway River. This included all transects for both survey types. We also evaluated only those transects between Bear Creek and White Cap Creek, as this river section has the most transects of both survey types.

Hook-and-line survey

Analysis of hook-and-line surveys was conducted on the available data for WCT (1975 - 2018) and RBT (1997 - 2018). Rainbow Trout data was not recorded during these surveys until 1997. The relative abundance of fishes susceptible to hook-and-line fishing was assessed by calculating catch rates (fish/h) for all species combined and individual species. We also evaluated long-term trends in the number and size (mean TL) of WCT and number of RBT caught through least squares regression as described above. Survey year was the independent variable and $\log_e$ transformed catch rates and mean TL were dependent variables (Maxell 1999; Kennedy and Meyer 2015).

RESULTS

SNORKEL SURVEY

Westslope Cutthroat Trout

1-person transects

In 2018, WCT were observed in 83% of the 1-person transects (Table 14 and Figure 37). The highest mean linear density of WCT was observed in the most downstream river section between Three-Links Creek and Race Creek (Table 15). The mean WCT linear density in 1-person transects was 5.8 fish/transect, which was the lower than long-term mean (11.0 fish/transect) and the lowest that has been observed since 2004 (Table 15 and Figure 38). Despite this, long-term (1973 - 2018) linear density has remained about the same, as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 16).

The mean linear density of WCT > 305 mm observed for 1-person transects was 5.3 fish/transect, which was higher than the longer term average of 2.2 fish/transect (Table 17 and Figure 38). There has been a stable long-term (1973 - 2018) trend in the linear density of WCT > 305 mm as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 16).

GPM main-stem transects

Westslope Cutthroat Trout were observed in all GMP main-stem transects (Table 14 and Figure 37). The highest densities of WCT were observed in the most upstream river section above White Cap Creek (Figure 37). The mean WCT density in GPM main-stem transects was 0.66 fish/100 m^2 (Table 14), which was equal to the long-term (1992 - 2018) mean areal density (0.66/100 m^2; Figure 39). There has been a stable long-term (1992 - 2018) trend in mean density, as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 18).

GPM tributary transects

Westslope Cutthroat Trout were observed in 88% of the GPM tributary transects (Table 11 and Figure 37). The highest densities of WCT were observed in Marten Cr. and Deep Cr. (Figure 37). The mean density of WCT in tributary GPM transects was 1.20/100 m^2 (Table 14), which was lower than the long-term (1988 - 2018) mean density (1.56/100 m^2; Figure 39). There has been a stable long-term (1988 - 2018) trend in mean density as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 18).

Rainbow Trout

1-person transects

In 2018, RBT were observed in 92% of the 1-person transects that were snorkeled in 2018 (Table 14 and Figure 40). The highest mean linear density of RBT were observed between Moose Cr. and Three-links Cr. (Figure 40). The mean RBT linear density for all 1-person transects was 12.9/transect, higher than the long-term mean linear density (7.7/transect) for surveys conducted since 1973 (Figure 41). However, there was a statistically significant declining trend in mean linear density from 1973 to 2017 (Table 18).

The mean linear density of RBT > 305 mm observed for 1-person transects was 0.04/transect (Figure 41). Linear density of RBT > 305 mm has declined since 2013; however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in mean linear density of RBT > 305 mm from 2002 to 2018 (Table 16).

GPM main-stem transects

Rainbow Trout were observed all GPM main-stem transects (Table 14 and Figure 40). The highest densities of RBT were observed above White Cap Cr. (Figure 40). The mean RBT density in GPM main-stem transects was 0.35/100 m^2 (Table 14), which was lower than the long-term mean density (0.51/100 m^2) for surveys conducted since 1992 (Figure 42). Additionally, there was a statistically significant declining trend (r_{intr} = -0.063) in RBT density in GPM main-stem transects from 1992 to 2018 (Table 18).

GPM tributary transects

Rainbow Trout were observed in every GPM tributary transect except one (Table 14 and Figure 40). The highest areal densities of RBT were observed in Marten Cr. and Three-links Cr. (Figure 40). The mean RBT areal density in GPM tributary transects was 1.88/100 m^2 (Table 14), which was approximately 50% of the long-term (1988 - 2018) mean density (3.73/100 m^2; Figure 42). Additionally, there was a statistically significant declining trend (r_{intr} = -0.032) in RBT mean density in GPM tributary transects from 1988 to 2018 (Table 18).

Mountain Whitefish

1-person transects

In 2018, MWF were observed in 67% of the 1-person transects (Table 14 and Figure 43). The highest mean linear density of MWF was observed between Three-links Cr. and Race Cr. (Figure 43). The mean MWF linear density in 1-person transects was 7.0 fish/transect, the lowest for any survey conducted since 1973 (Table 14 and Figure 44). Additionally, there was a statistically significant long-term declining trend (r_{intr} = -0.049) in MWF mean linear density from 1973 to 2018 (Table 16).

The mean linear density of MWF > 305 mm observed for 1-person transects was 4.2 fish/transect, the lowest since 2009 (Figure 44). There was not a significant long-term trend (2002 - 2018) in mean linear density of MWF > 305 mm as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 16).

GPM main-stem transects

Mountain Whitefish were observed in all GPM main-stem transects (Table 14 and Figure 43). The highest densities of MWF were observed above White Cap Cr. (Figure 43). The mean MWF areal density in GPM main-stem transects was 0.53/100 m^2 (Table 14), which was approximately 50% of the long-term mean density (1.02/100 m^2) for surveys conducted since 1992 (Figure 45). However, there was not a significant long-term trend (1992 - 2018) in mean density of MWF as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 18).

GPM tributary transects

Mountain Whitefish were observed in 63% of the GPM tributary transects (Table 14 and Figure 43). The highest densities of MWF were observed in Bear Cr. Lower and Moose Cr. #1 transects (Figure 43). The mean MWF density in GPM tributary transects was 0.48/100 m^2 (Table 14), which was lower than the long-term mean density (0.62/100 m^2) for surveys conducted since 1988 (Figure 45). Additionally, there was a statistically significant declining trend (r_{intr} = -0.029) in MWF density in GPM tributary transects from 1992 to 2018 (Table 18).

Evaluation of snorkel survey methods

No correlation ($R^2 < 0.0$) was detected when we compared annual mean linear densities of 1-person surveys to annual mean densities of GPM sites (Figure 50). A further comparison using only those survey transects between Bear and White Cap creeks showed some correlation (R^2 = 0.28) although this correlation was influenced by one data point (Figure 50).

Hook-and-line survey

An estimated 112 angler hours resulted in the catch of 269 WCT, 103 RBT, 1 MWF, and 1 WCT x RBT hybrid. The average catch rate (for all species) of 3.3 fish/h in 2018 was higher than the average for surveys conducted since 2012 (3.1 fish/h; Table 19). The average catch rate for WCT of 2.4 fish/h in 2018 was equal to the average for surveys conducted since 2012 (Table 19). Westslope Cutthroat Trout catch in 2018 (n = 269) was 25% lower than the long-term average of 356 fish (Figure 46); however, there has been a stable trend (90% error bounds around estimates of r_{intr} overlapped zero) in WCT CPUE during hook-and-line surveys conducted from 2012 to 2018 (Table 20). The mean TL of WCT caught in 2018 (278 mm) was the highest since 2010, and was larger than the long-term average of 260 mm for hook-and-line surveys conducted from 1975 to 2018 (Figure 47). There was a statistically significant increasing trend ($r_{intr} > 0$) in mean TL of WCT caught during hook-and-line surveys conducted from 1975 to 2018 (Table 20). The percent of WCT > 305 mm caught was the highest of any survey conducted since 1975 (Figure 48). There was a statistically significant increasing trend (r_{intr} = 0.016) in percent of WCT > 305 mm caught during hook-and-line surveys conducted from 1975 to 2018 (Table 20).

The number of RBT caught by hook-and-line in 2018 (n = 103) was higher than the long-term average of 77 (Figure 49). However, there has been a stable trend in RBT CPUE during hook-and-line surveys conducted from 2012 to 2018 (Table 20).

DISCUSSION

SNORKEL SURVEY

Westslope Cutthroat Trout

Although abundance of WCT fluctuates from year to year, there has been no significant trend in WCT linear or density in the main-stem Selway River since 1973. There was also no significant trend in density of WCT > 305 mm during this time period. This contrasts with increasing population trends (total abundance and fish > 305 mm) observed in the North Fork Clearwater, St. Joe, and Coeur d'Alene rivers (Hand et al. 2020; Ryan et al. 2020). In the St. Joe and Coeur d'Alene rivers, increases were primarily attributed to regulation changes and favorable weather conditions, with improvements in habitat and water quality contributing to a lesser extent (Ryan et al. 2020). Similar to these other north Idaho Rivers, WCT density in the Selway River more than doubled after restrictive regulations were implemented in 1976 (Lindland 1977b). Since then, abundances have fluctuated around the long-term mean. The lack of a significant trend could be because this population was not as depressed as the others due to its remoteness. With the Selway River located in a Wilderness area with limited access, minimal anthropogenic impacts, and restrictive regulations, environmental factors and observer variability likely explain the annual fluctuation in WCT densities reported in these snorkel surveys.

Similar to the main-stem Selway River, there has been no significant long-term trend in WCT density in tributaries of the Selway River since 1988. Overall, it appears that the WCT population in the Selway River basin is stable for fish of all sizes and only those > 305 mm, and therefore meets our management objectives of providing a high quality fishery. However, we observed the highest % of WCT > 305 mm along with lower overall abundance. This suggests that there has been poor recruitment in recent years, and that the population may experience declines over the next few years.

Rainbow Trout

There has been a significant declining trend in RBT density in the main-stem Selway River since 1973. A similar trend was observed in the Lochsa River and MFSR (Messner and Schoby 2019; *See* Lochsa River section in this report). This decline occurred prior to 2000, and may be partially explained by the cessation of stocking RBT in the Selway River in 1990. From 1968 to 1990, ~5,000 RBT were stocked into the Selway River annually. Additionally, adult steelhead returns across Idaho have been declining (Dobos et al. 2020). The low densities in the mid-late 1990's correspond to the lowest wild steelhead counts observed at Lower Granite Dam. Since many of the RBT observed in the Selway River are likely juvenile and residualized steelhead, the trends in RBT may be driven just as much by adult steelhead returns and juvenile survival as other factors.

There was also a significant declining trend in RBT density in the tributaries of the Selway River since 1988. As was observed in the main-stem river, tributary densities declined in the late 1990s. A similar trend was observed in the Lochsa River and MFSR (Messner and Schoby 2019; *See* Lochsa River section in this report). The low wild adult steelhead returns in the 1990s appears to have impacted densities of RBT in tributaries as well as the main-stem Selway River.

Few RBT observed in the Selway River are > 305 mm. The lack of larger RBT is likely because juvenile steelhead migrate before reaching this size, and this population is dominated by steelhead as opposed to resident fish. With few larger fish, they are not targeted heavily by anglers. Thus, angling and regulation changes have likely had little impact on trends in the RBT population in the Selway River.

Mountain Whitefish

There was a significant declining trend in MWF linear density in the main-stem Selway River since 1973, while neither the GPM main-stem nor tributary transects showed a significant trend in density. Declines in MWF have been observed in the main-stem of other northern Idaho rivers, including the Lochsa and South Fork Clearwater River (*See* Lochsa River and SFCR chapters in this report). Long-term declines in MWF populations have been documented in other locations across the southern portion of their range as well, including the Big Lost River and

Kootenai River, Idaho, the Yampa River, Colorado, and the Madison River, Montana (Paragamian 2002; IDFG 2007; Boyer 2016). While the direct cause has not been identified, the declines in these systems coincided with occurrences of low flows and higher water temperatures (Brinkman et al. 2013). These studies also suggested that habitat alteration, irrigation, nonnative fish interactions, disease, and harvest are also likely contributing to declines in MWF populations (IDFG 2007; Boyer 2016). Some of these factors generally do not apply to the wilderness of the Selway River drainage; however, increases in water temperature, disease, observer variability, and survey timing could be potential factors.

Increased water temperatures could affect MWF populations in several ways. Warmer temperatures may impact populations through increased mortality (Jager et al. 1999; Copeland and Meyer 2011; Kennedy and Meyer 2015). Mean monthly summer air temperatures have been above normal every year except one since 1996 (NOAA 2021). Additionally, severe outbreaks of Proliferative Kidney Disease (PKD) have been observed in Montana, and the disease is known to be present in Idaho (Phillips 2016; Hutchins et al. 2021). While no major fish kills have been directly observed, minor die-offs have been observed in in Idaho rivers during summer months. Thus, PKD could impact populations through lower level mortality. As such, increased water temperatures could impact MWF movement, survival, and recruitment before some other species.

Sampling biases may also be playing a part in the recent decline in main-stem density. In 2005, we began conducting our snorkel surveys based on river flow instead of a set calendar date. Surveys now tend to occur at higher flow levels, which could reduce visibility of a bottom-dwelling fish compared to previous surveys. This coincides with the decline in linear density observed from 2003 to 2018, and may be contributing to the lower linear densities observed during this time frame.

In contrast to the decline in MWF linear density in the main-stem Selway River, there was no significant trend in linear density of MWF > 305 mm. This would suggest that the trend in this MWF population is size-dependent, with declines occurring at smaller sizes. However, there could be more observer variability with smaller fish. Within the SFCR, fewer juvenile MWF have been observed during snorkel surveys over the last 10 years compared to historic surveys (*IDFG unpublished data*; Putnam et.al 2017*; Scott Putnam, personal communication*). No MWF < 150 mm have been recorded during surveys of the main-stem river below White Cap Creek and no more than three have been observed in any year above White Cap Creek. Fish populations are often limited by recruitment, and changes in juvenile survival would have long-lasting effects on the population (Bradford and Cabana 1997; Pope et al. 2010). If changes in habitat or temperature regimes are occurring, a decline in juvenile abundance may be an early indicator, and would explain why we are seeing declining trends in the overall population.

The trends observed in MWF density in the Selway River are likely a combination of changes in our sampling strategy and potential disease/environmental factors affecting survival and recruitment. Regardless, the recent downward trend in MWF densities across northern Idaho rivers and other parts of their historic range warrants a more thorough evaluation.

EVALUATION OF SNORKEL SURVEY METHODS

Comparisons of the 1-person and GPM snorkel survey data sets for WCT in the main-stem Selway River showed little to no correlations between years. These results pose a problem for effectively evaluating trends in the fish populations in the Selway River drainage. Differences in the locations and number of survey transects between the two methods likely explains for some of this. There are 28 1-person survey transects spread throughout the main-stem Selway River (Race Creek to White Cap Creek). In contrast, there are nine main-stem GPM transects located primarily upstream of Bear Creek (7 of 9), while four are upstream of White Cap Creek where there are no 1-person transects. Thus, transects snorkeled in the two surveys differ in river width, discharge, and water temperatures, and we would expect to see annual differences in fish distribution. A comparison of data for the St. Joe River also shows differing long-term trends when comparing all transects to only transects in the upper reaches (above Prospector Creek; IDFG *unpublished data*). The higher correlation (R^2) observed in the Selway River when we compared the two methods only in transects between Bear Creek and White Cap Creek suggests that the differences in site distribution is at least partly responsible for the differing population trends. Another contributing factor is likely due to differences in survey techniques between the two survey methods. This is supported by the much larger variability in annual mean density that occurs in the 1-person transects. One-person surveys potentially incur larger observation error based on discharge, visibility, observer experience/bias, and position within a transect with only one snorkeler. In contrast, GPM surveys employ more people to cover the entire width of the stream, and may have less observer bias as an inexperienced surveyor would have less overall impact (Apperson et al. 2015). However, it can be difficult to maintain proper spacing and communicate during fast moving down-stream snorkel surveys. This may introduce bias not encountered in 1-person surveys. Due to the long-term nature of both survey methods, it is not prudent to eliminate either survey type. At this time, the GPM transects may be a more accurate technique. We recommend using caution while drawing conclusions from the 1-person transects, as they are more likely to only show large-scale changes in fish density. In the future, we recommend attempting to reduce as much observer bias as possible by maintaining consistent sampling practices and ensuring snorkelers are well trained. Additionally, we recommend evaluating observer bias in 1-person surveys by having 2-3 different snorkelers replicate each 1-person transect (Zubik et al 1988; Rodgers et al. 1992; Thurow 1994). This should provide insight into variability based on different snorkelers and experience level.

HOOK-AND-LINE SURVEY

There has been a significant increasing trend in WCT average TL and percent of fish > 305 mm caught by hook-and-line for surveys conducted since 1973. The catch rate of 3.3 fish/h for hook-and-line surveys in 2018 was similar to the average for surveys conducted in the Selway River from 2012 to 2018, but at the lower end of the range (2.8 - 5.8) of catch rates on float trips conducted on the MFSR (Messner and Schoby 2019). However, the catch rate for WCT (2.4/h) was similar to the MFSR in 2016 and 2017 (Messner and Schoby 2019). The percent of WCT > 305 mm caught is similar to those reported for Middle Fork Salmon River (MFSR) float trips (Messner and Schoby 2019). In contrast to the increasing trend in the Selway River, this proportion has remained stable in the MFSR. The stable annual catch, long-term increasing trends in TL of WCT caught by hook-and-line, and increasing percent of WCT > 305 mm observed in the Selway River indicate we are meeting our management goals of providing a high-quality fishery with abundant larger fish. However, as mentioned previously, the higher percentage of larger fish may indicate that there has been poor recruitment in recent years, and that the population may experience declines over the next few years.

There was no significant trend in annual catch of RBT during float trips on the Selway River. This aligns with the lack of a trend in RBT abundance in 1-person snorkel transects during similar time periods (mid-1990’s to present). As discussed previously, this is likely attributable to the cessation of RBT stocking in the Selway River in 1990.

MANAGEMENT RECOMMENDATIONS

1. Continue to conduct Selway River snorkel and hook-and-line surveys to monitor trends in WCT, RBT, and MWF abundance and size structure.

2. Attempt to minimize observer bias in snorkel surveys by maintaining consistent sampling practices and ensuring snorkelers are well trained. Evaluate observer bias in 1-person surveys by having 2 - 3 different people snorkel each 1-person transect individually at 10-minute intervals.

Table 14. Densities (fish/100 m^2; GPM transects) and linear densities (fish/transect; 1-person transects), by transect, for snorkel surveys in the Selway River drainage, Idaho, in 2018.

GPM transects

					Areal density					
River section	Transect name	Transect length (m)	Transect width (m)	Temp °C	Westslope Cutthroat Trout	RBT	Mountain Whitefish	Chinook Salmon	Trout fry	Bull Trout
Tributaries	Bear Creek Lower	59	22	20.0	1.09	3.65	1.71	0.16	0.00	0.08
	Bear Creek Upper	96	23	19.5	0.00	0.00	0.00	0.00	0.00	0.00
	Deep Cr, Cactus	73	8	17.0	2.94	4.90	0.00	0.00	0.00	0.00
	Deep Cr, Scimitar	111	7	17.0	1.11	4.08	0.00	0.00	0.00	0.12
	Little Clearwater, #1	42	15	15.0	0.48	1.13	0.64	0.32	0.32	0.00
	Little Clearwater, #2	54	11	15.0	1.45	2.58	0.00	0.00	0.32	0.00
	Marten Creek	13	8	14.0	5.13	18.46	0.00	0.00	0.00	0.00
	Moose Creek #1	53	24	18.5	1.18	2.44	2.28	0.00	0.00	0.00
	Moose Creek, East Fork #2	44	31	17.0	1.10	0.44	0.00	0.29	0.00	0.00
	Moose Creek, East Fork #3	65	16	16.0	1.63	2.21	0.10	0.00	0.00	0.00
	Moose Creek, North Fork	83	25	14.5	0.67	0.43	0.58	0.00	0.00	0.00
	Running Creek #1	42	14	17.0	1.87	1.19	0.00	0.17	0.00	0.00
	Running Creek #2	68	12	18.0	0.00	0.98	0.00	0.00	0.00	0.00
	Three Links Creek #1	30	8	14.5	0.83	12.50	0.00	0.00	0.00	0.00
	White Cap, Strata 3, #1	94	15	20.0	0.49	3.52	1.20	0.00	0.00	0.00
	White Cap, Strata 3, #2	99	17	17.0	0.24	1.80	0.24	0.00	0.00	0.00
	White Cap, Strata 3, #3	107	17	15.0	0.22	0.22	0.05	0.00	0.00	0.00
	Tributary mean density				1.20	3.56	0.40	0.06	0.04	0.01
	90% CI				0.34	1.35	0.21	0.20	0.03	0.01
Above White Cap Cr.	Hell's Half Acre	73	15	12.0	0.54	0.54	1.26	0.45	0.18	0.00
	Magruder Crossing	173	21	14.0	0.71	0.55	0.68	0.52	0.05	0.00
	Beaver Point	169	14	16.0	0.69	0.04	0.48	2.25	0.13	0.04
	Little Clearwater	73	16	16.0	1.34	1.43	1.59	2.35	0.00	0.08
Running Cr. to Bear Cr.	Badluck Cr	78	43	17.5	0.59	0.06	0.53	0.00	0.00	0.00
	Big Bend	103	40	15.0	0.29	0.17	0.14	0.29	0.00	0.00
	Northstar	118	38	16.0	0.55	0.04	0.88	0.00	0.00	0.00
Bear Cr. to Moose Cr.	Osprey Island	125	38	15.0	0.95	0.27	0.69	0.00	0.00	0.00
Moose Cr. to Three-links Cr.	Below Tango	152	51	15.5	0.31	0.09	0.10	0.00	0.00	0.00
	Main-stem mean density				0.66	0.35	0.71	0.65	0.04	0.01
	90% CI				0.34	1.35	0.21	0.20	0.03	0.01
	All transects mean density				1.02	2.45	0.51	0.26	0.04	0.01
	90% CI				0.34	1.35	0.21	0.20	0.03	0.01

1-person transects

					Linear density					
River section	Transect name	Transect length (m)	Transect width (m)	Temp °C	Westslope Cutthroat Trout	RBT	Mountain Whitefish	Chinook Salmon	Trout fry	Bull Trout
White Cap Cr. to Runnning Cr.	1/2 Mile Below White Cap	62	15	-	1	3	0	0	0	0
	1 mile Below White Cap	81	31	-	0	1	3	0	0	0
	Cougar Bluff	66	16	-	1	5	0	0	0	0
Running Cr. to Bear Cr.	1/2 mile Below Running	54	27	-	5	7	26	0	0	1
	Archer	52	25	-	2	2	5	0	0	0
	Above Goat Creek Rapid	94	30	-	10	1	1	0	0	0
	Selway Lodge	75	23	-	5	5	12	0	0	0
Bear Cr. to Moose Cr.	Above Rodeo	54	23	-	9	15	0	0	0	0
	Below Rodeo	95	15	-	11	30	10	0	0	0
	Below Pettibone	80	32	-	3	20	12	0	0	0
	Rattlesnake Bar	130	42	-	1	1	1	0	0	0
	Below Ham	110	25	-	3	8	1	0	0	0
	Below Hell Creek	130	26	-	5	3	0	0	0	0
Moose Cr. to Three-Links Cr.	Moose Creek Confluence	55	28	-	3	14	15	0	0	0
	Divide Creek	100	31	-	0	30	2	0	0	0
	Above Ladle	120	35	-	11	67	0	0	0	0
	Below Ladle	85	31	-	4	28	0	0	0	0
	Below Osprey Rapid	90	24	-	5	7	0	0	0	0
Three-Links Cr. to Race Cr.	Below 3-links	150	31	-	0	3	2	0	0	0
	Dry Bar	50	28	-	12	55	31	0	0	0
	Above Wolf Creek	150	47	-	2	1	7	0	0	0
	Above Renshaw	80	51	-	3	0	0	0	0	0
	Otter	25	10	-	44	0	39	2	0	0
	Packer	65	42	-	0	3	1	0	0	0
	Mean density				5.8	12.9	7.0	0.1	0.0	0.0
	90% CI				3.0	5.9	3.6	0.1	---	0.1

Table 15. Average linear density (fish/transect) of all sizes of Westslope Cutthroat Trout (WCT) in sections of the main-stem Selway River, Idaho, determined by 1-person snorkel surveys from 1973 to 2018.

All WCT

	River section					
Year	White Cap Creek to Running Creek	Running Creek to Bear Creek	Bear Creek to Moose Creek	Moose Creek to Three-Links Creek	Three Links Creek Race Creek	Mean
1973	4.2	7.2	5.3	4.5	5.0	5.2
1974	3.4	4.8	7.5	8.2	3.4	5.5
1975	6.8	6.6	5.0	6.3	4.6	5.9
1976	7.2	6.2	6.0	8.8	6.1	6.9
1977	10.8	18.6	17.4	22.0	9.3	15.6
1978	7.4	10.6	19.6	20.9	9.8	13.6
1980	13.2	18.6	16.0	21.7	17.2	17.3
1982	11.2	11.2	16.2	20.3	20.8	15.9
1984	11.0	17.4	19.4	25.7	16.3	17.9
1986	15.2	19.2	21.4	26.1	24.6	21.3
1988	13.3	11.6	21.8	24.3	17.4	17.7
1990	6.8	16.4	7.4	6.8	11.7	9.8
1992	4.8	9.4	6.2	4.4	3.0	5.6
1994	7.5	9.0	8.3	3.0	6.0	6.8
1995	13.0	13.3	13.3	6.0	6.4	10.4
1996	10.7	15.5	15.0	8.5	30.0	15.9
1997	6.0	26.5	7.8	10.5	15.0	13.2
1998	---	---	1.0	2.0	7.6	---
1999	17.0	12.6	16.6	10.6	4.2	12.2
2001	13.3	12.7	7.5	5.3	1.3	6.2
2002	12.7	21.0	8.6	12.6	2.2	9.8
2003	10.3	8.3	10.6	---	---	---
2004	8.0	5.0	7.0	12.0	5.5	5.6
2005	13.5	6.0	8.4	19.8	6.7	10.1
2007	2.3	4.5	3.6	1.8	15.3	6.3
2008	15.3	8.5	15.0	14.8	10.3	10.6
2009	6.7	4.0	10.2	21.4	12.5	11.2
2010	7.0	9.0	13.8	31.3	16.0	13.1
2011	11.5	10.2	12.5	22.8	11.0	10.0
2012	5.3	8.8	5.8	9.2	8.8	7.8
2013	4.7	16.2	8.5	52.8	11.2	15.0
2015	8.0	12.0	17.2	13.3	10.6	12.8
2017	6.3	15.0	6.4	19.6	15.7	13.2
2018	0.7	5.5	5.3	4.6	10.2	5.8

Table 16. Intrinsic rate of change (r_{intr}) in linear density (fish/transect) for Westslope Cutthroat Trout (WCT), Rainbow Trout (RBT), and Mountain Whitefish (MWF) observed in 1-person snorkel surveys conducted in the main-stem Selway River, Idaho, from 1973 to 2018. Significance was set at α = 0.10. Data sets include fish of all sizes unless specified as "> 305 mm", which only includes fishes of this size.

1-person

Species	Data set	r_{intr} estimate	90% CI lower	90% CI upper
WCT	All fish	0.002	-0.007	0.012
	> 305 mm	0.008	-0.009	0.025
RBT	All fish			
	1973-2018	-0.049	-0.066	-0.032
	> 305 mm			
	2002-2018	0.239	-0.044	0.521
MWF	All fish			
	1973-2018	-0.014	-0.024	-0.005
	> 305 mm			
	2002-2018	-0.059	-0.119	0.001

Table 17. Average linear density (fish/transect) of Westslope Cutthroat Trout (WCT) > 305 mm in sections of the main-stem Selway River, Idaho, determined by 1-person snorkel surveys from 1973 to 2018.

WCT > 305 mm

	River section					
Year	White Cap Creek to Running Creek	Running Creek to Bear Creek	Bear Creek to Moose Creek	Moose Creek to Three-Links Creek	Three Links Creek Race Creek	Mean
1973	0.4	0.8	1.8	0.6	1.2	1.0
1974	0.6	0.4	1.2	1.3	0.3	0.7
1975	0.8	1.2	0.4	0.7	1.4	0.9
1976	1.6	1.0	1.5	1.9	1.2	1.4
1977	2.4	4.0	4.4	3.3	2.5	3.3
1978	1.2	2.2	4.2	3.1	3.0	2.7
1980	1.7	2.2	1.6	3.9	1.8	2.2
1982	1.0	1.2	2.4	4.2	3.5	2.5
1984	1.7	3.6	4.4	6.2	4.8	4.1
1986	3.2	2.8	4.0	5.9	3.6	3.9
1988	3.3	2.6	5.0	5.8	3.2	4.0
1990	2.0	2.6	1.2	1.4	3.7	2.2
1992	0.3	2.4	3.0	0.3	0.9	1.4
1994	0.5	1.0	1.0	0.0	0.0	0.5
1995	0.0	0.0	1.2	0.3	0.0	0.3
1996	0.0	0.0	0.0	0.9	3.0	0.8
1997	0.5	3.0	1.5	1.6	2.3	1.8
1998	---	---	0.0	0.3	3.0	---
1999	1.0	0.3	2.8	3.4	1.0	1.7
2001	3.7	5.0	4.2	2.3	0.7	3.2
2002	5.7	4.3	2.3	2.2	0.8	3.1
2003	0.7	1.7	3.3	---	---	---
2004	1.0	1.0	1.4	1.7	0.5	1.1
2005	4.0	1.0	0.9	3.4	1.8	2.2
2007	0.3	2.0	0.7	0.0	5.0	1.6
2008	2.7	1.8	2.0	0.8	0.8	1.6
2009	0.0	0.3	0.8	2.8	4.3	1.6
2010	1.7	2.8	2.0	2.3	4.0	2.5
2011	3.8	4.7	3.8	7.0	1.3	4.1
2012	0.3	0.3	0.2	3.0	2.2	1.2
2013	3.0	7.8	2.0	2.8	1.8	3.5
2015	1.0	0.8	2.7	2.3	3.8	2.1
2017	1.0	0.8	1.2	3.0	0.3	1.3
2018	0.7	5.5	5.3	4.6	10.2	5.3

Table 18. Intrinsic rate of change (r_{intr}) in areal density (fish/100 m^2) for Westslope Cutthroat Trout (WCT), Rainbow Trout (RBT), and Mountain Whitefish (MWF) observed in snorkel surveys conducted in the main-stem (1992 - 2018) and tributaries (1988 - 2018) of the Selway River, Idaho. Significance was set at α = 0.10.

GPM

Species	Data set	r_{intr} estimate	90% CI lower	90% CI upper
WCT	Tributaries	0.020	-0.004	0.043
	Main-stem	0.004	-0.017	0.025
RBT	Tributaries	-0.032	-0.055	-0.009
	Main-stem	-0.063	-0.103	-0.024
MWF	Tributaries	-0.029	-0.049	-0.008
	Main-stem	0.012	-0.009	0.032

Table 19. Effort (h) and catch rates (fish/h) for all fishes combined (overall) and only Westslope Cutthroat Trout (WCT) for hook-and-line surveys of the Selway River, Idaho, from 2012 to 2018.

Year	Effort	Catch rate Overall	Catch rate WCT
2012	155	3.4	2.5
2013	140	2.7	2.2
2015	140	3.3	2.5
2017	135	3.0	2.6
2018	112	3.3	2.4
Mean	136	3.1	2.4

Table 20. Intrinsic rate of change (r_{intr}) in CPUE (fish/h), mean total length, and proportion of fish > 305 mm for Westslope Cutthroat Trout (WCT; 1975 - 2018) and Rainbow Trout (RBT; 1997 - 2018) caught during hook-and-line surveys in the Selway River, Idaho. Significance was set at α = 0.10.

Species	Data set	r_{intr} estimate	90% CI lower	90% CI upper
WCT	CPUE	0.008	-0.024	0.040
	Mean length	0.002	0.001	0.002
	> 305 mm	0.016	0.010	0.021
RBT	CPUE	-0.017	-0.215	0.180

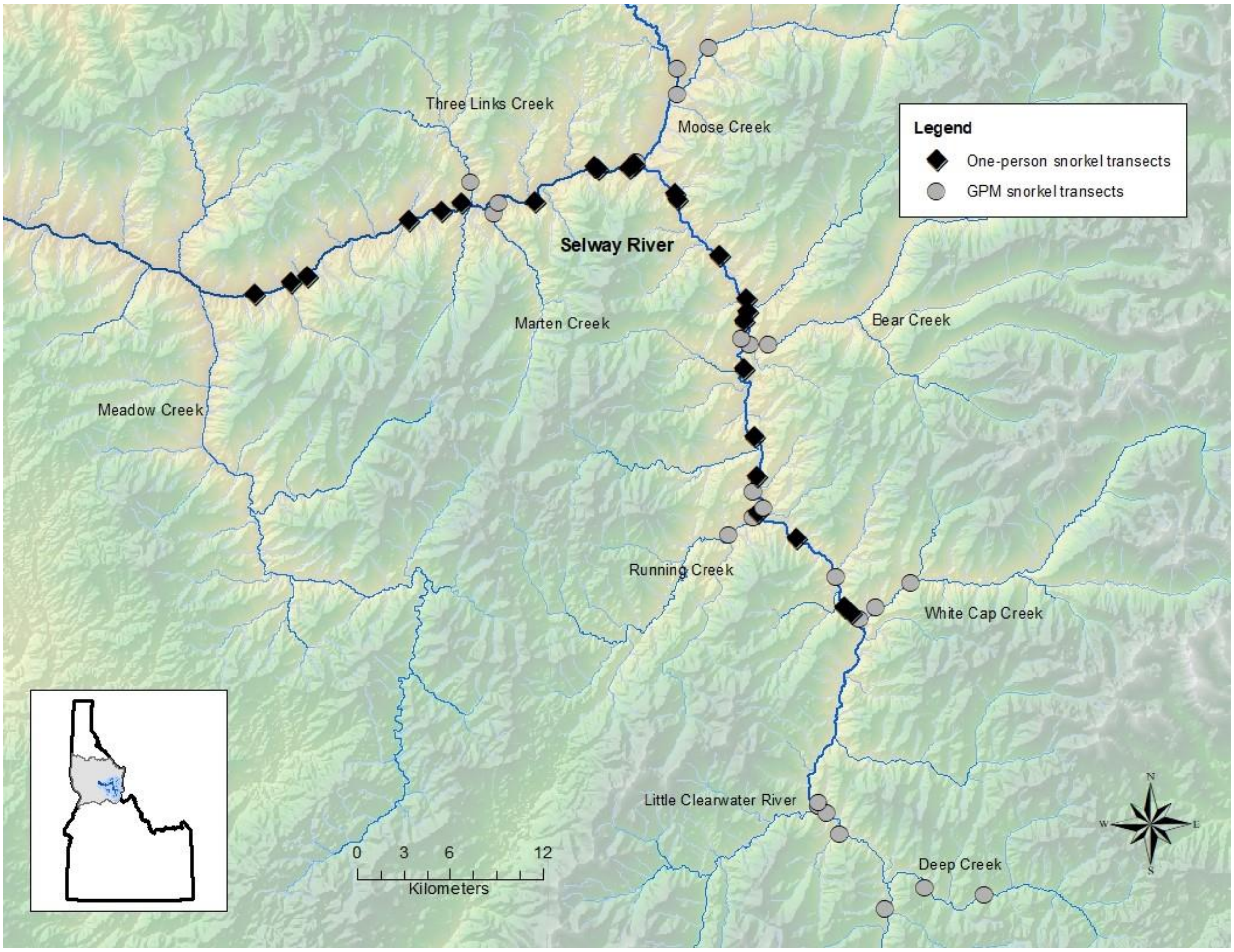

Figure 36. Map showing locations of General Parr Monitoring (GPM) and 1-person snorkel transects surveyed in the Selway River basin, Idaho, in 2018.

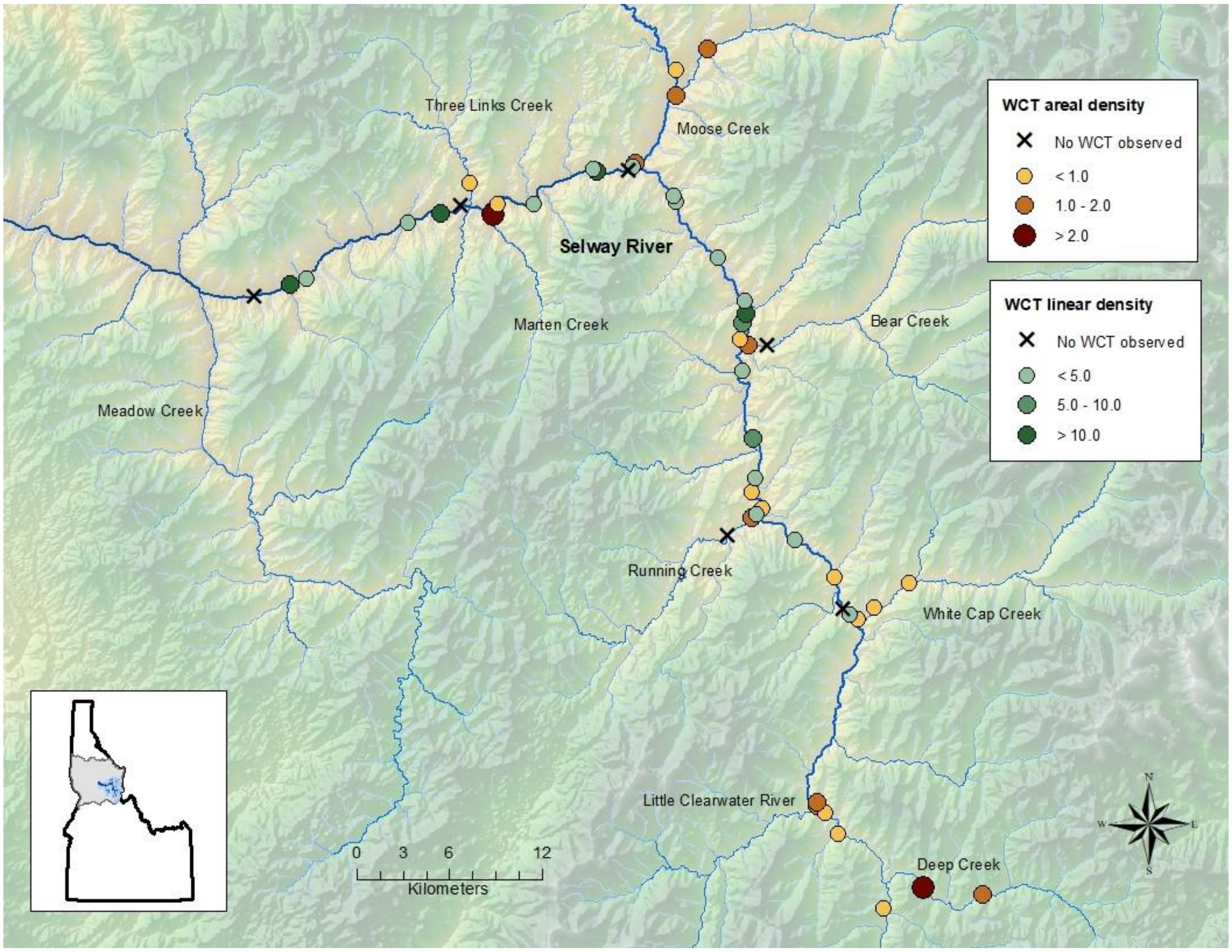

Figure 37. Westslope Cutthroat Trout (WCT) linear density (fish/100 m^2) in General Parr Monitoring transects and areal density (fish/transect) in 1-person transects observed for each snorkel transect surveyed in the Selway River basin, Idaho, in 2018.

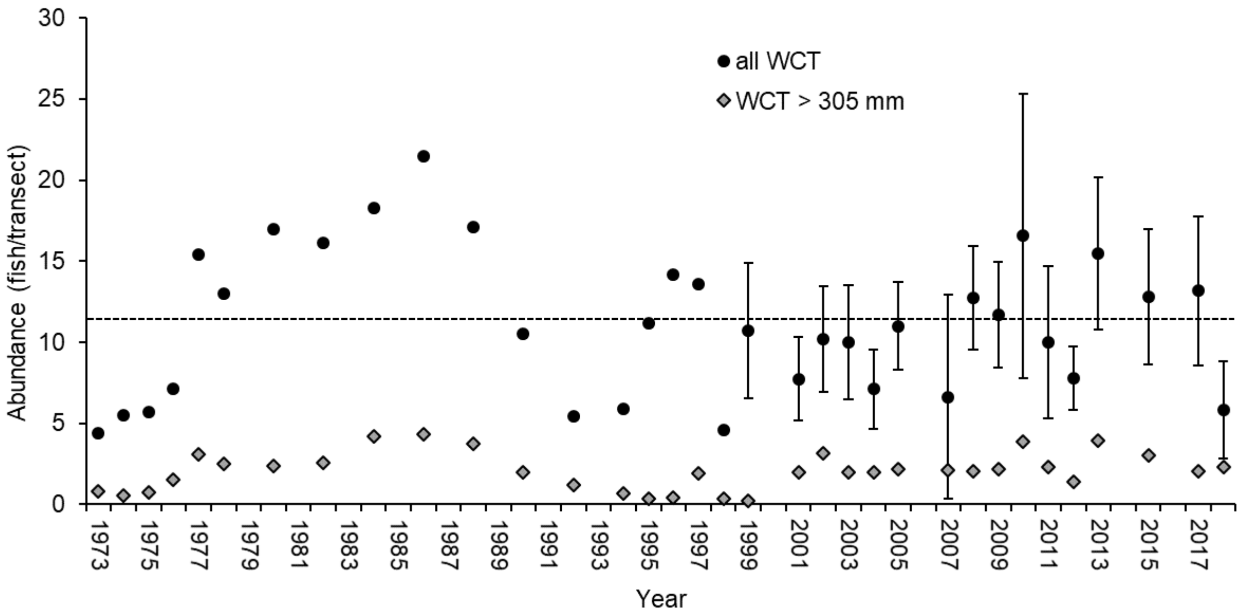

Figure 38. Mean linear density (all sizes and only fish > 305 mm) of Westslope Cutthroat Trout (WCT) observed in 1-person snorkel transects in the main-stem Selway River, Idaho, from 1973 to 2018. The dashed line represents the mean abundance for all sizes of WCT observed across all years surveyed. Error bars represent 90% CIs.

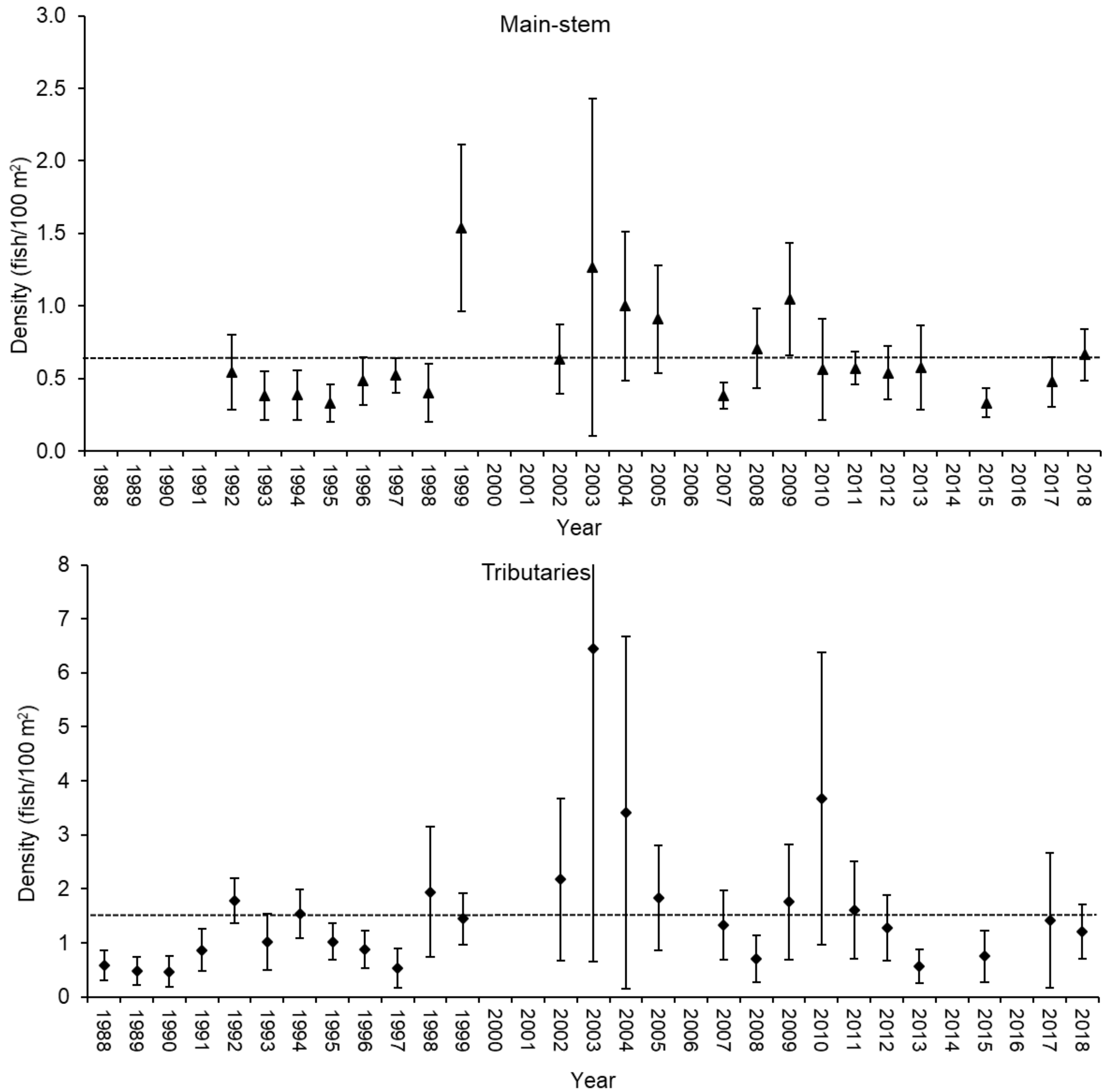

Figure 39. Mean densities of Westslope Cutthroat Trout (WCT) observed in General Parr Monitoring snorkel transects in the main-stem (1992 - 2018) and tributaries (1988 - 2018) of the Selway River, Idaho. The dashed lines represent the mean densities of WCT across all years surveyed. Error bars represent 90% CIs.

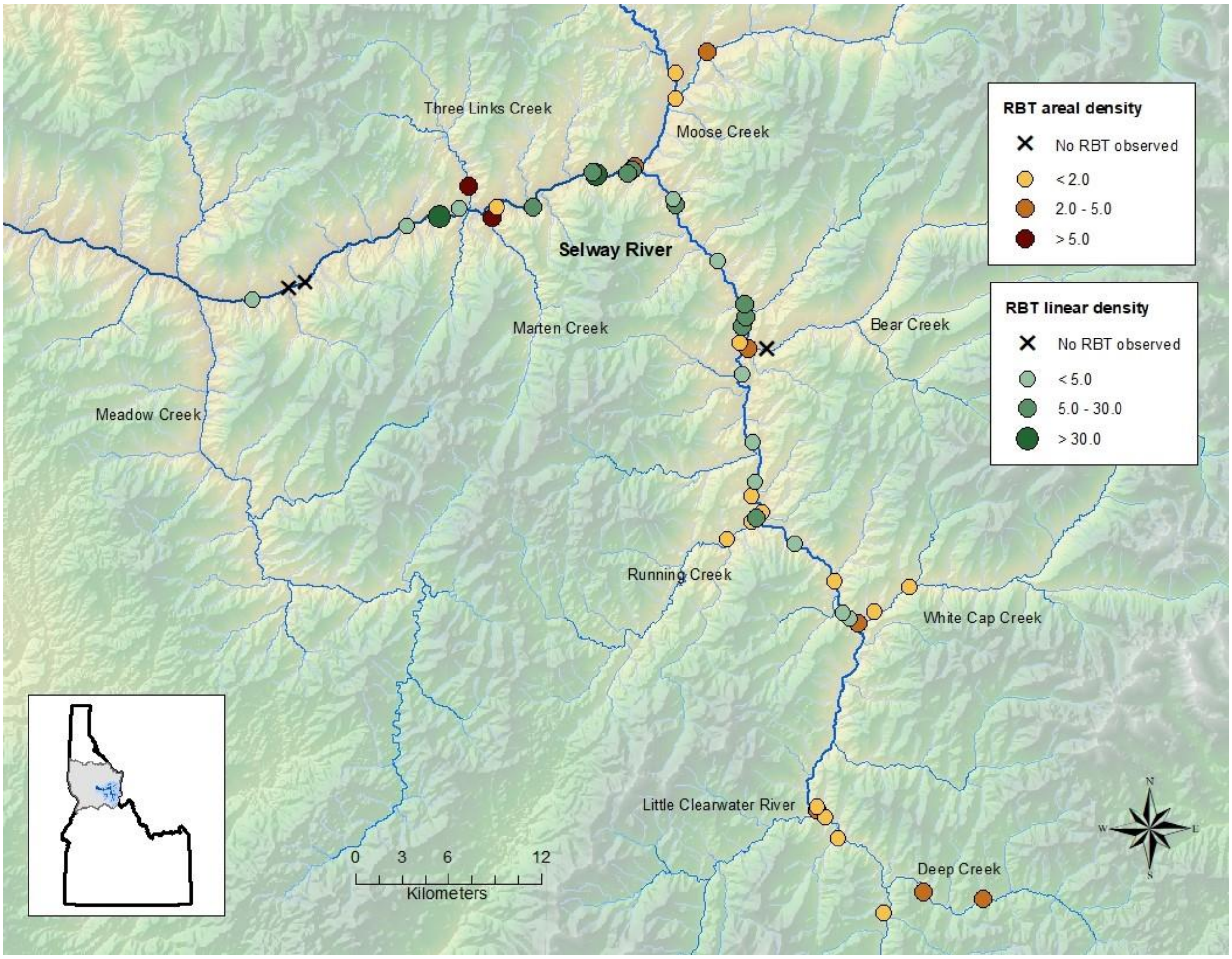

Figure 40. Rainbow trout (RBT) density (fish/100 m^2) in General Parr Monitoring transects and linear density (fish/transect) in 1-person transects observed for each snorkel transect surveyed in the Selway River basin, Idaho, in 2018.

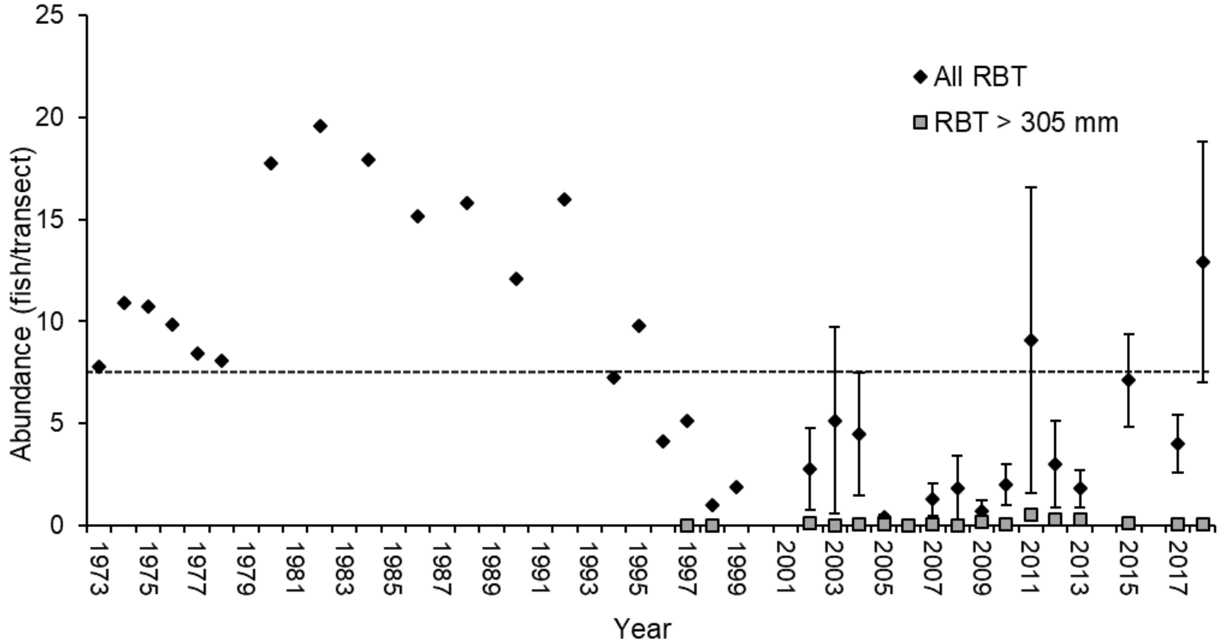

Figure 41. Mean linear density (all sizes and only fish > 305 mm) of Rainbow Trout (RBT) observed in 1-person snorkel transects in the main-stem Selway River, Idaho, from 1973 to 2018. The dashed line represent the mean abundance of all sizes of RBT observed across all years surveyed. Error bars represent 90% CIs.

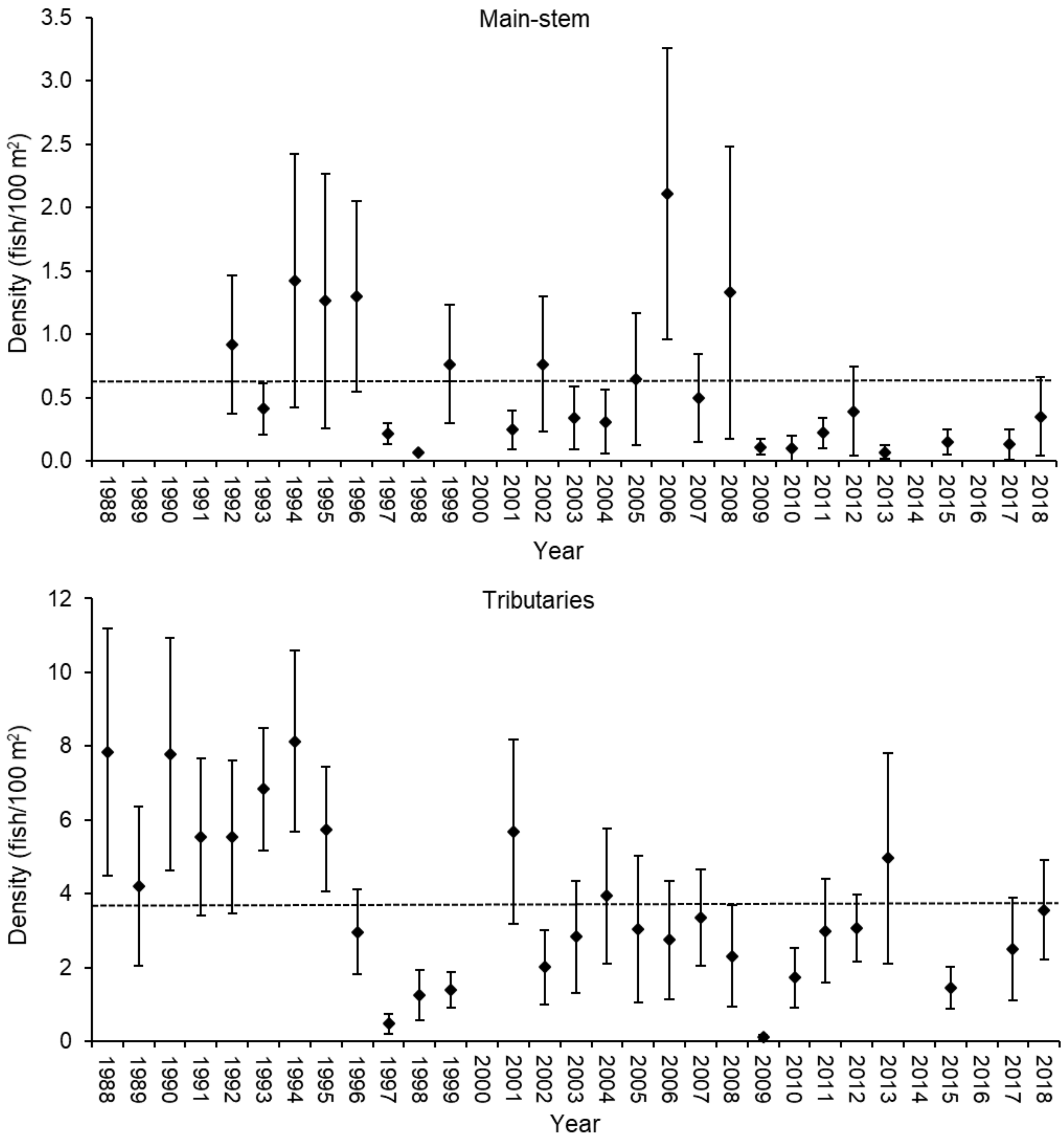

Figure 42. Mean areal density of Rainbow Trout observed in General Parr Monitoring snorkel transects in the main-stem (1992 - 2018) and tributaries (1988 - 2018) of the Selway River drainage, Idaho. The dashed lines represents the mean densities of Rainbow Trout across all years surveyed. Error bars represent 90% CIs.

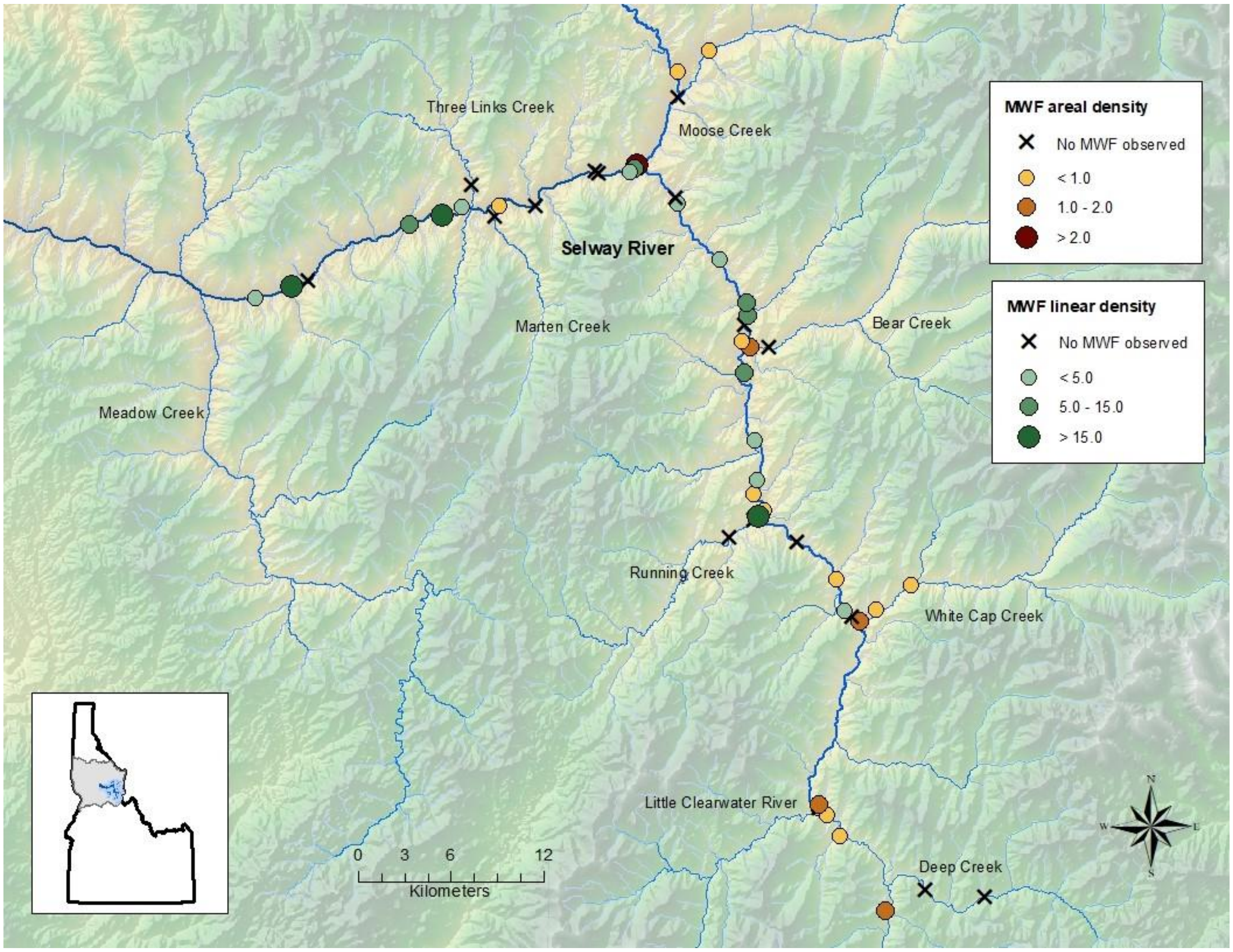

Figure 43. Mountain Whitefish density (fish/100 m^2) in General Parr Monitoring transects and linear density (fish/transect) in 1-person transects observed for each snorkel transect surveyed in the Selway River basin, Idaho, in 2018.

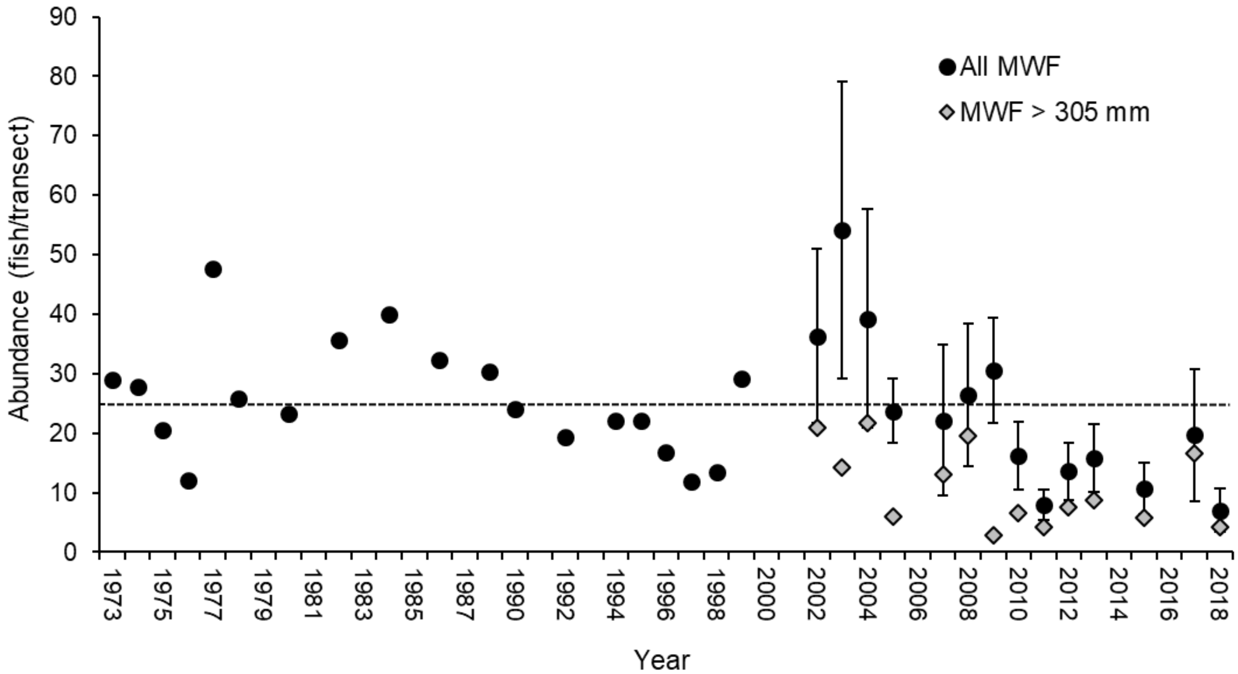

Figure 44. Mean linear density (all sizes and only fish > 305 mm) of Mountain Whitefish (MWF) observed in 1-person snorkel transects in the main-stem Selway River, Idaho, from 1973 to 2018. The dashed line represents the mean abundance of MWF across all years surveyed. Error bars represent 90% CIs.

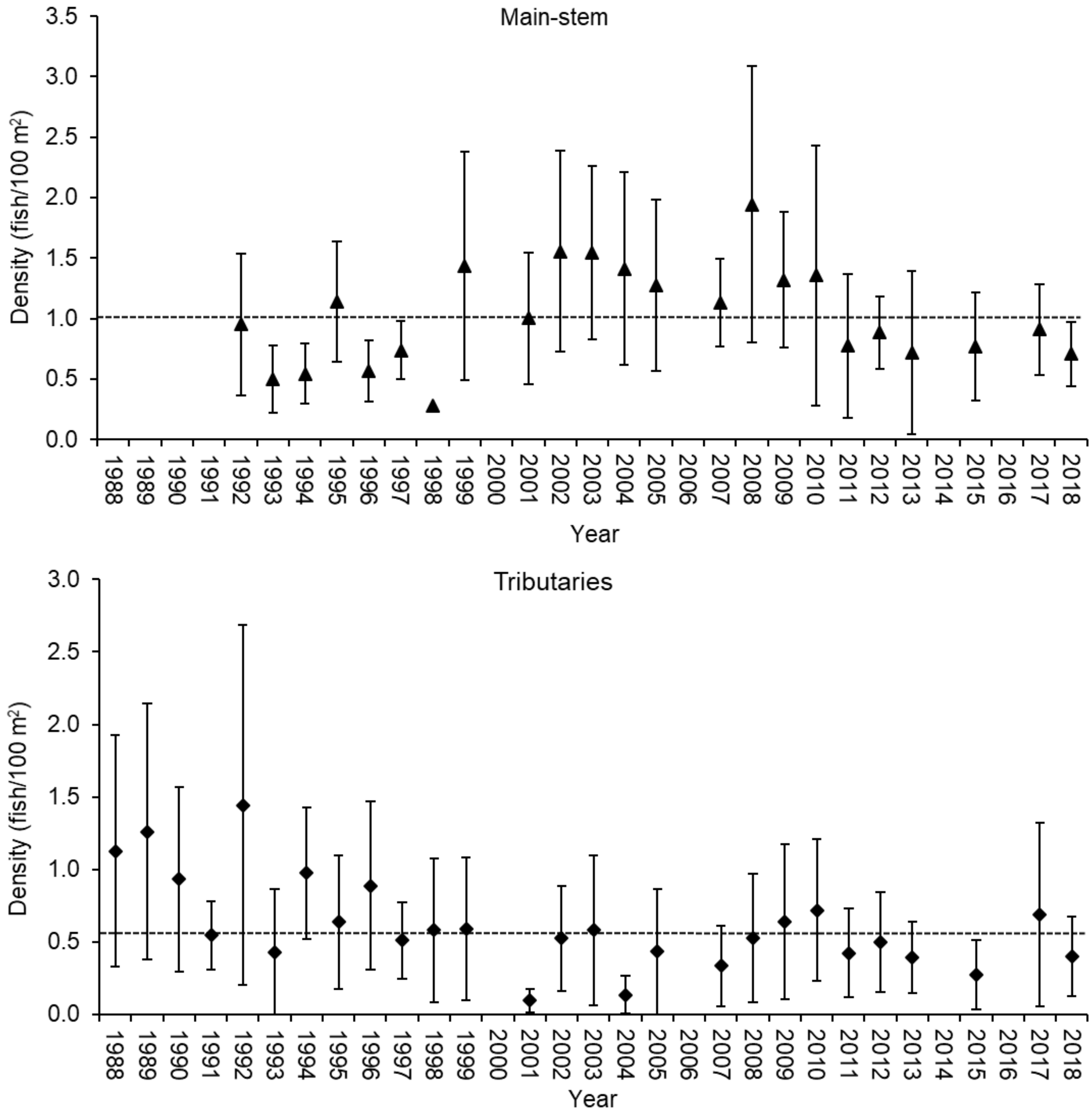

Figure 45. Mean density of Mountain Whitefish observed in General Parr Monitoring snorkel transects in the main-stem (1992 - 2018) and tributaries (1988 - 2018) of the main-stem Selway River, Idaho. The dashed lines represent the mean densities of Mountain Whitefish observed across all years surveyed. Error bars represent 90% CIs.

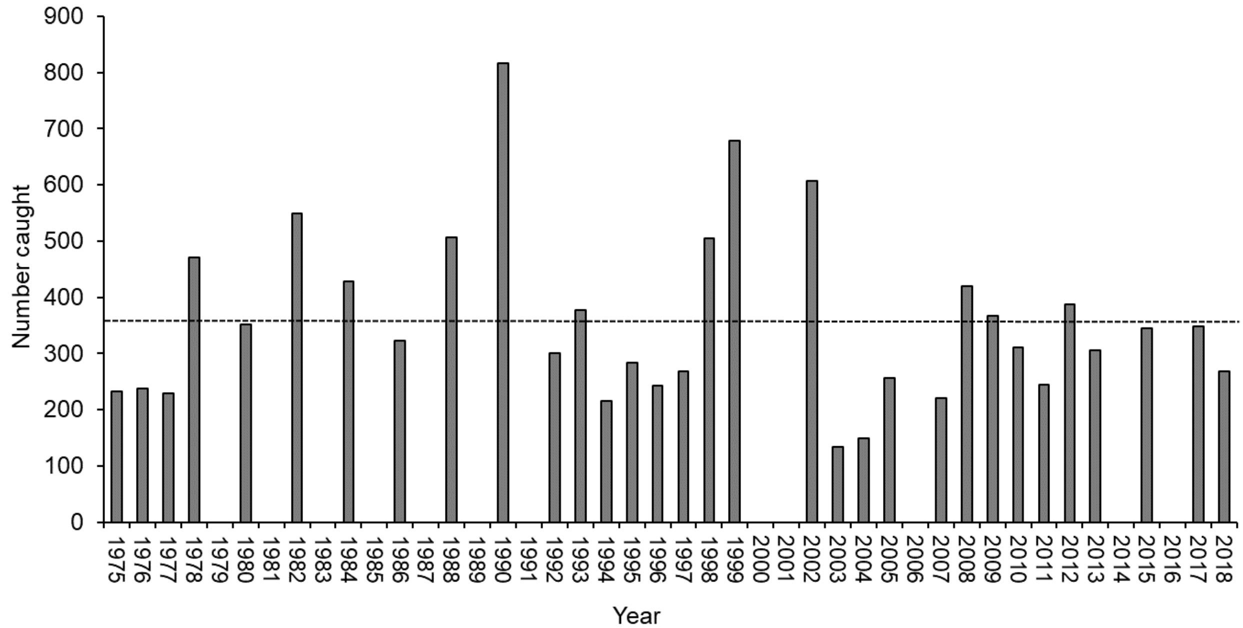

Figure 46. Number of Westslope Cutthroat Trout caught by hook-and-line surveys in the Selway River, Idaho, from 1975 to 2018.

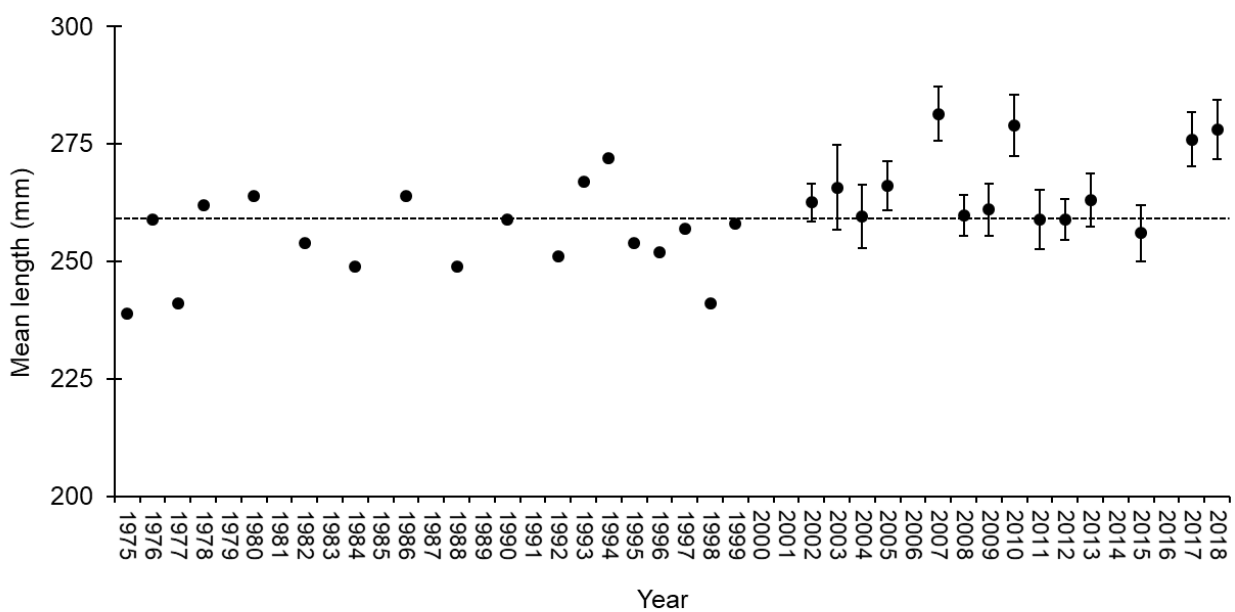

Figure 47. Mean total length of Westslope Cutthroat Trout caught by hook-and-line surveys in the Selway River, Idaho, from 1975 to 2018. The dashed line represents the average length of Westslope Cutthroat Trout observed across all years surveyed. Error bars represent 90% CIs.

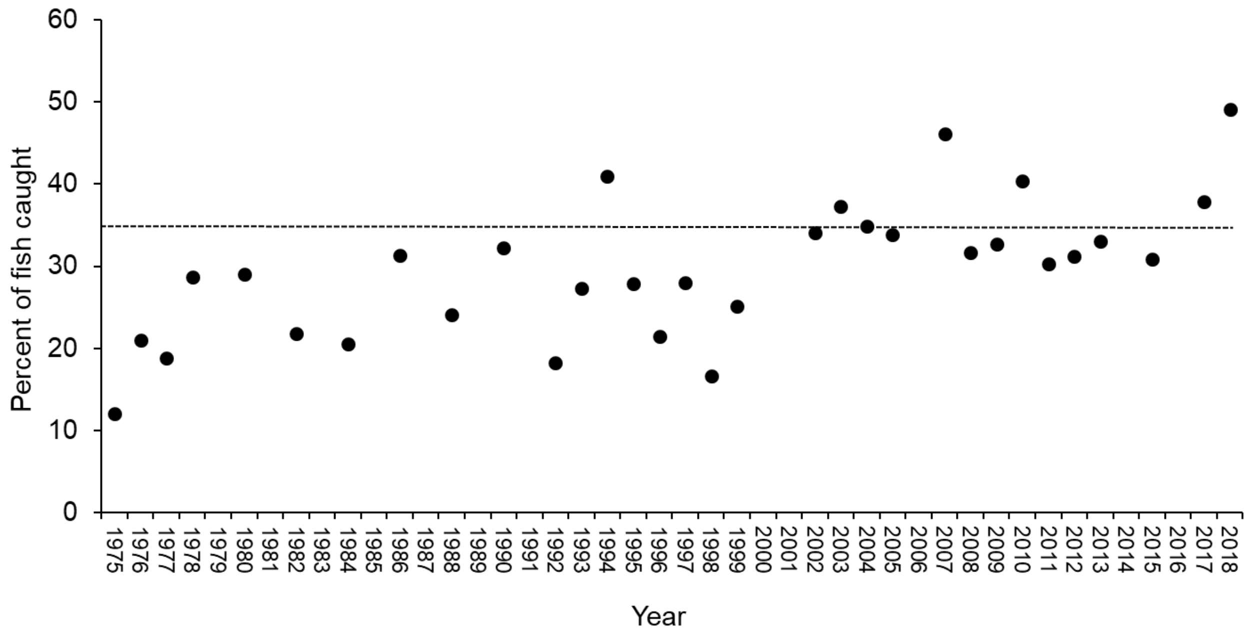

Figure 48. Percent of Westslope Cutthroat Trout > 305 mm caught by hook-and-line sampling in the Selway River, Idaho, from 1975 to 2018. The dashed line represents the mean percent of fish caught > 305 mm across all years surveyed.

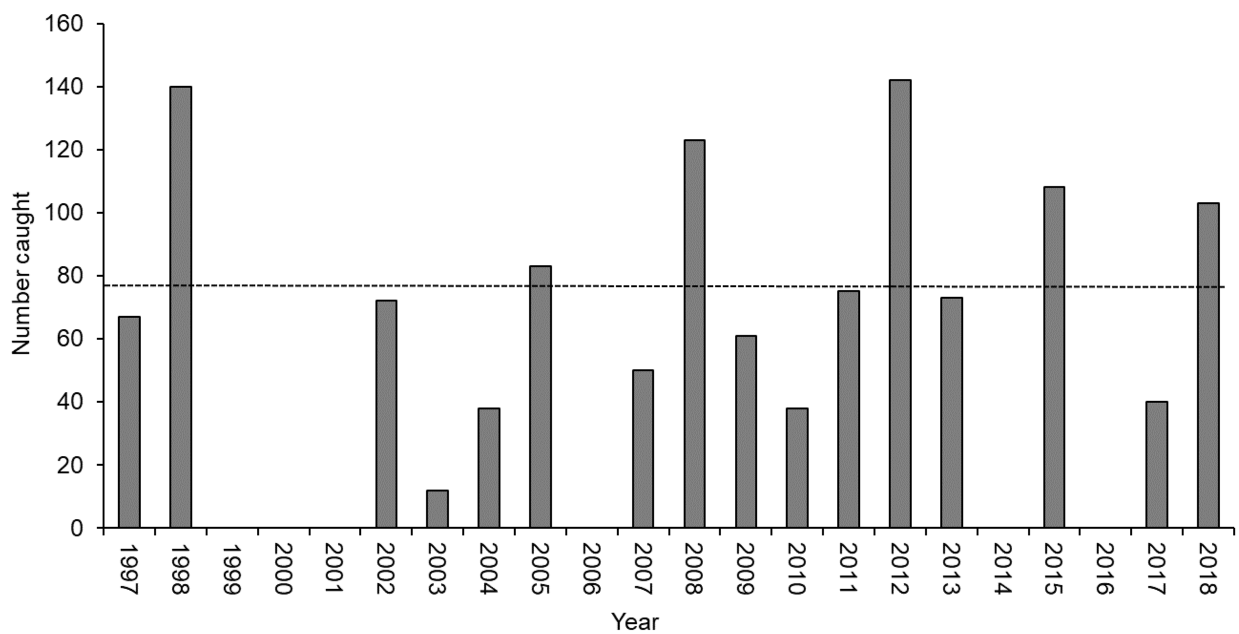

Figure 49. Number of Rainbow Trout caught by hook-and-line surveys in the Selway River, Idaho, from 1997 to 2018. The dashed line represents the mean number of fish caught across all years surveyed.

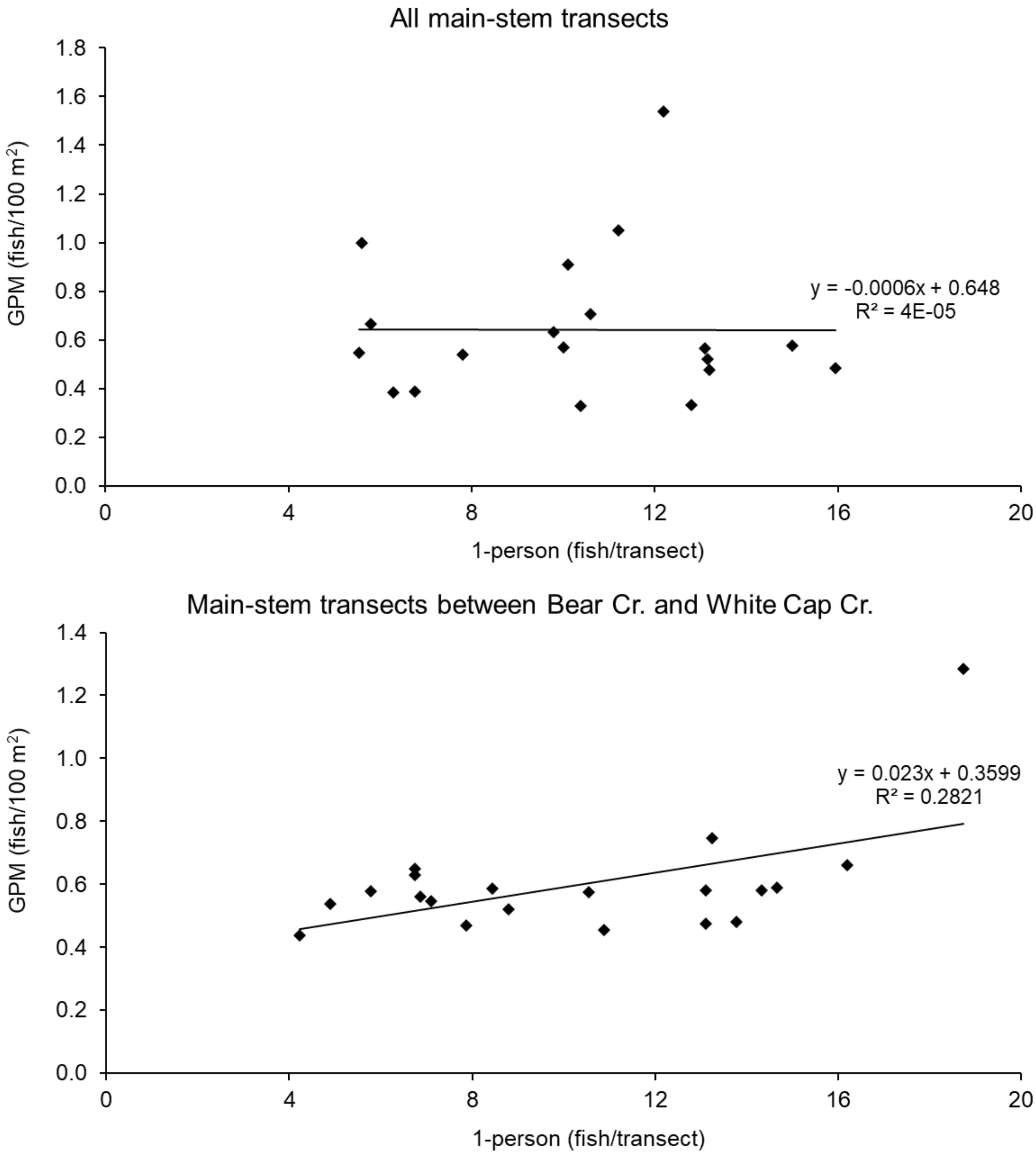

Figure 50. Comparison of Westslope Cutthroat Trout linear density (fish/transect) in 1-person transects and areal density (fish/100 m^2) in General Parr Monitoring transects on the main-stem Selway River, Idaho, from 1992 to 2018 (years with both data sets), for all main-stem transects, and only those transects occurring between Bear Creek and White Cap Cr.

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EVALUATION OF FISH POPULATIONS IN THE SOUTH FORK CLEARWATER RIVER

ABSTRACT

Snorkel surveys were conducted on the main-stem South Fork Clearwater River in 2018 to assess trends in Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT), Rainbow Trout *O. mykiss* (RBT), and Mountain Whitefish *Prosopium williamsoni* (MWF) abundance and size structure after implementing catch-and-release regulations for WCT in 2011. No significant trend in mean WCT density (all fish and just those > 305 mm) was observed for surveys conducted from 2000 to 2018. While RBT density was higher than those seen in the Lochsa and Selway river systems, there was no significant trend in mean RBT density. For MWF, there was a significant declining trend in density of all size combined, but not for just those > 305 mm. Four Smallmouth Bass *Micropterus dolomieu* were observed. The lack of an increasing trend and overall low densities of WCT in the SFCR are likely due to potential illegal harvest, poor habitat, and higher water temperatures found in this river system,. The higher densities of RBT observed in the SFCR was likely due to its warmer temperatures and higher productivity compared to the Lochsa River and Selway River. While the direct cause of the decline in MWF populations is unknown, factors such as higher summer temperatures, prevalence of winter anchor ice, and disease (such as Proliferative Kidney Disease) may be impacting populations. With no significant trend in density of MWF > 305 mm, it appears the population decline is primarily occurring in smaller sizes. Smallmouth Bass were not observed in the main-stem SFCR sampling until 2014. Since then, they have been observed in low abundance with no evidence of reproduction.

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INTRODUCTION

Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* (WCT) are distributed throughout the South Fork Clearwater River (SFCR) drainage, occupying both the main-stem river and tributaries. Both resident and fluvial life history forms are present (Dobos 2015). While WCT are abundant in the other major Clearwater River tributaries (Lochsa, Selway, and North Fork Clearwater rivers), they are generally found in much lower densities in the SFCR (Cochnauer et al. 2001; Schill et al. 2005; Dobos 2015). This is mostly believed to be due to the higher water temperatures found in the SFCR system compared to the other tributaries (Dobos 2015). Fish populations in the SFCR have been regularly evaluated through snorkel surveys for a variety of projects. However, the main-stem river has only been surveyed three times since 2000. In 2011, daily bag limits on the main-stem SFCR were changed from two WCT (none < 356 mm) to a trout limit of six (all must have clipped adipose fin). This regulation change was implemented to protect WCT, and provide opportunity for anglers to keep hatchery Rainbow Trout *O. mykiss* (RBT) and residualized hatchery steelhead stocked in the SFCR basin. To evaluate impacts of this regulation change, and track long-term trends in abundance and distribution of SFCR resident fisheries, we have initiated a four-year sampling rotation to survey these main-stem transects on a two years on, two years off basis.

OBJECTIVES

1. Determine whether the density and size structure of Westslope Cutthroat Trout in the South Fork Clearwater River have changed since implementing catch-and-release regulations in 2011.

2. Evaluate trends in the density and size structure of Rainbow Trout and Mountain Whitefish *Prosopium williamsoni* (MWF) in the South Fork Clearwater River.

3. Evaluate whether the Smallmouth Bass *Micropterus dolomieu* (SMB) distribution is expanding in the South Fork Clearwater River.

STUDY AREA

Snorkel surveys were conducted on the main-stem SFCR, located in Idaho County, Idaho (Figure 51). The SFCR has a total drainage area of approximately 302,130 ha. Approximately 69% is located on National Forest lands, 23% is private ownership, and the remaining 8% is owned by other state and federal agencies, and the Nez Perce Tribe (Dobos 2015). Elevation of the main-stem SFCR ranges from 378 to 1,186 m. Mean discharge ranges from is 6 m^3/s in September to 90 m^3/s in May.

METHODS

FIELD SAMPLING

A snorkel survey was conducted on the main-stem SFCR from July 29 to 31, 2018. A total of 21 snorkel transects were surveyed (Figure 51). To maintain consistency, we surveyed the same transects as previous surveys conducted from 2000 to 2018 (Hand et al. 2021). These are a subset of transects surveyed as part of the Idaho Supplementation Studies program, developed to evaluate supplementation as a potential tool for recovery of Snake River basin Chinook Salmon *O. tshawytscha* (Lutch et al. 2003). Transects were surveyed using the standard snorkeling methodologies outlined in Apperson et al. (2015). This technique entails snorkeling downstream using an appropriate number of snorkelers to cover the entire width of the river to allow for the calculation of fish densities. All fish observed were counted, and total length (TL) was estimated to the nearest inch for all game species. We do not record presence of adipose fin clips for RBT, as it is generally too difficult to observe accurately. Non-game species (e.g. *Cottus* spp, *Catostomus* spp.) were categorized as > or < 305 mm. Transect length (m) and average width (m; based on five measurements) was measured using a Nikon ProStaff S laser rangefinder. Visibility (m) was estimated at each transect by holding a Keson 50-m, reel-style, fiberglass measuring tape underwater. A snorkeler backed away from the reel until lettering was indistinguishable, then moved back towards the reel until the lettering was viewable again. The distance from snorkeler to the reel was recorded. Habitat type, date, time of day, water temperature, and weather conditions were also recorded for each transect. Juvenile steelhead and resident RBT are indistinguishable and are collectively referred to as "RBT". This report focuses on WCT, RBT, and MWF. Results and analysis of data collected on Chinook Salmon in 2018 can be found in Roth et al. (2019).

DATA ANALYSIS

We calculated density (fish/100 m^2) of WCT, RBT, and MWF observed in each transect by dividing the total number of fish observed by the area snorkeled. Annual mean density was calculated by averaging the density of all transects snorkeled in a year. Data was summarized by both individual transect and by river section. We divided the SFCR into three river sections based on geomorphic differences as described by Dobos (2015): Lower - mouth to river kilometer (RKM) 30.0; Middle - RKM 30.0 to RKM 75.0; Upper - upstream of RKM 75.0. The Middle section consists of steep canyons and higher gradient compared to the Lower and Upper sections which were characterized as unconfined with large floodplains and lower gradients. We used least squares regressions to evaluate trends (2000 - 2018) in mean density of WCT, RBT, and MWF of all sizes across all transects, and by river section. We also assessed trends (2000-2018) in size structure by evaluating mean density of WCT and MWF > 305 mm using least squares regression. All regression analysis used survey year (2000 - 2018) as the independent variable and $\log_e$ transformed density as dependent variables (Maxell 1999; Kennedy and Meyer 2015). The intrinsic rate of change in the population (r_{intr}) was determined by the slope of the regression line fit to these data. A 90% CI was calculated for r_{intr} to determine significance, where the trend is considered significant when $r_{intr} \neq 0$ and the error bounds do not include 0. We used a significance level of $\alpha = 0.10$.

For RBT, we developed a length-frequency distribution to assess size structure. To help assess what percent of the observed RBT were wild, the length frequency graph of observed fish was compared to a length frequency of steelhead smolts that were stocked into the SFCR in March, 2018 (a subset of these fish were measured by Clearwater Hatchery staff at the time of stocking). Spatial distributions of WCT, RBT, and MWF were visually represented by plotting mean density for each transect on maps of the survey area using GIS software.

RESULTS

The game fishes observed in the 2018 snorkel survey included RBT (n = 145), Chinook Salmon (n = 22), WCT (n = 9), MWF (n = 147), and SMB (n = 4) (Table 21). Water temperatures ranged from 24°C in the most downstream transects to 16°C in one transect in the Upper River section (Table 17).

WESTSLOPE CUTTHROAT TROUT

Westslope Cutthroat Trout were observed in five transects in 2018 (Table 12 and Figure 52). No more than four WCT were observed in any transect. Mean density (0.02 fish/100 m^2) was the lowest observed of the four surveys conducted since 2000 (Table 22). There was not a significant long-term trend (2000 - 2018) in the mean density for all sizes of WCT as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 23). The mean density of WCT > 305 mm (0.003/100 m^2) was similar to 2017 (0.004/100 m^2).

Mean densities of WCT in each river section were similar to 2017 (Table 22). Mean density has remained near 0 in the lower river section since 2000, but has declined in the middle and upper river sections since 2010 (Table 22). However, there were no significant long-term trends (2000 - 2018) in mean density of WCT in any river section, as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 24). There was no significant long-term trend (2000 - 2018) in the mean density of WCT > 305 mm as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 23). The mean TL of WCT observed in 2018 was the highest of any survey (Table 25).

RAINBOW TROUT

Rainbow Trout were observed in 57% of transects snorkeled in 2018 (Table 26). The highest densities were observed in the middle river section, especially between Johns Creek and Newsome Creek (Figure 53). The mean RBT density was the lowest for any survey conducted since 2000 (Table 26); however, there was no significant long-term trend (2000 - 2018) in the mean density of RBT as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 23). Mean density of RBT declined in all three river sections from 2017 to 2018, with the largest decline occurred in upper river section (Table 26). There was a significant declining long-term trend (2000 - 2018) in the mean density of RBT in the upper river section, while there was no significant long-term trend in the other river sections (Table 24).

The mean RBT TL observed in 2018 was 153 mm, similar to 2000 and 2010, but nearly 60% larger than the mean TL observed in 2014 and 2017 (92 mm; Table 27). The RBT observed during August snorkel surveys tend to have a smaller length distribution than the steelhead stocked into the SFCR in April (Figure 54). About 3% of the RBT observed in the snorkel surveys were > 279 mm, compared to < 1% of the stocked steelhead.

MOUNTAIN WHITEFISH

Mountain Whitefish were observed in 62% of transects snorkeled in 2018, with the highest densities occurring in the middle river section (Table 22 and Figure 55). The mean density of MWF observed in 2018 was higher than what was observed in 2017; however, there was a significant declining long-term (2000 - 2018) trend in density (Table 23). Mean MWF density in each river section increased from 2017 to 2018, but were still lower than most other years. There were significant declining long-term trends (2000 - 2018) in the mean density of MWF in the upper and middle river sections, while there was no significant long-term trend in the lower river section (Table 24).

The mean density of MWF > 305 mm (0.24/100 m^2) was three times higher than 2017; however, there was no significant long-term trend (2000-2018) in the mean density of MWF > 305 mm as the 90% error bounds around estimates of r_{intr} overlapped zero (Table 23).

SMALLMOUTH BASS

Four SMB (150 - 300 mm) were observed in one transect (18.2 KM) in 2018, similar to 2014 (3.5 KM) and 2017 (18.2 KM) when they were also observed in only on transect.

DISCUSSION

WESTSLOPE CUTTHROAT TROUT

Implementing catch-and-release regulations in 2011 has not increased WCT densities in the SFCR. Additionally, no change size structure of WCT has been observed. This contrasts with significant increases in WCT density and size structure being observed in the main-stem Selway River, Lochsa River, Kelly Creek, and St. Joe River after catch-and-release regulation were implemented in those rivers (Johnson and Bjornn 1975; Ryan et al. 2020; Hand et al. 2021). In these rivers, changes in abundance and size structure were observed within a year or two after catch-and-release limits were implemented. This suggests that harvest is not currently limiting WCT abundance, and that other factors such as habitat conditions are more likely limiting WCT abundance in the SFCR. Conservation officers have suggested that illegal harvest could be high in the SFCR. This may partially stem from a lack of knowledge about fishing regulations. However, this is purely anecdotal and we do not have confirmed evidence of an illegal harvest issue.

Although there has not been an improvement in the size structure of WCT, small individuals from 25 to 150 mm have been observed during each survey. This indicates that there is at least some recruitment and survival occurring in the SFCR.

Based on the water temperatures observed in recent surveys, it is likely temperature and habitat issues play a large role in the low densities of WCT in the SFCR. The average temperature in the SFCR was 2 - 4°C warmer than the Lochsa, Selway, and St. Joe rivers in 2017 (Ryan et al. 2020; Hand et al. 2021). Higher temperatures are likely due to degraded riparian habitat from historic mining and timber harvest activities (Cochnauer and Claire 2001; Dechert and Woodruff 2003; Dobos 2015). Additionally, temperatures were detected that exceeded thermal tolerances (22°C) for adult WCT (Bjornn and Reiser 1991; Bear et al. 2007; Dobos 2015). No WCT were observed in transects where water temperatures > 21°C were recorded. Westslope Cutthroat Trout in the SFCR migrate long distances to seek out cooler temperatures in tributaries and upper reaches of the SFCR during the time of our survey (Dobos 2015). However, we did not observe many WCT in the Upper section of the SFCR where cooler temperatures were documented. Previous studies have suggested that the Upper River section contains poorer quality habitat (degraded riparian areas) than the Middle River section, which could account for its lower densities (Dobos 2015).

Enough time has passed since the implementation of catch-and-release regulations for WCT that improvements in the abundance and size structure of this fishery should have occurred if legal harvest were responsible for its depressed nature. With concern that illegal harvest and lack of knowledge about regulations could be impacting the population, we recommend increased enforcement presences and installing signs to inform anglers about the regulations. We recommend continuing to evaluate trends in density and size structure of WCT on a two year on, two year off basis, and explore opportunities to initiate or support habitat improvement projects that could increase densities in the SFCR drainage.

RAINBOW TROUT

The mean density of RBT observed in the SFCR has fluctuated between survey years resulting in no significant long-term trend since 2000. This is similar to trends in RBT density in the Lochsa and Selway Rivers since 2000 (*See* Lochsa River and Selway River sections of this report). The lack of a trend is likely a combination of a low number of survey years, and the timing of the declines in density observed in other rivers (mid-late 1990's), which occurred before our surveys in SFCR.

Rainbow Trout mean density in 2018 was higher in the SFCR (0.55 fish/100 m^2) than in the Selway River (0.35 fish/100 m^2) and Lochsa River (0.07 fish/100 m^2; *See* Selway River and Lochsa River chapters in this report). In contrast, it was lower than those typically observed in the Potlatch River (> 1.5 fish/100 m^2; Putnam et al. 2018). The densities we have observed in the SFCR could be explained by the annual stocking of hatchery steelhead smolts. Up to 17% of hatchery steelhead smolts may residualize and remain in a river system as resident fish (McMichael et al. 1997; Hausch and Melnychuk 2012). However, it is difficult to observe adipose fin clips during surveys, so we are unable to confirm what proportion of RBT observed are of hatchery origin. Since any residualized smolts would have grown for several months before our snorkel surveys, the majority of RBT observed in the SFCR snorkel surveys were likely smaller than the stocked smolts would have been at the time of our survey. This indicates that few of the RBT (most likely the larger individuals) observed were residualized hatchery smolts, and that the population is comprised primarily of resident RBT and naturally produced juvenile steelhead. Additionally, most hatchery smolts are released downstream of Meadow Creek, while the highest densities of RBT observed in our surveys were upstream of this area. Therefore, the higher densities of RBT observed in the SFCR may be due to its warmer temperatures and higher productivity compared to the Lochsa River and Selway River. This is supported by the presence of high densities of RBT in more productive systems like the Potlatch River (Putnam et al. 2018; Knoth et al. 2021). We recommend attempting to determine if the RBT observed during snorkel transects have clipped adipose fins. This will provide insight into what extent hatchery steelhead contribute to the resident fishery.

The mean TL of RBT observed in the SFCR (~153 mm) was similar to that observed in a 2017 snorkel survey of the Potlatch River basin (~144 mm), but higher than both the Selway River (~101 mm) and Lochsa River (~71 mm; Putnam et al. 2018; Knoth et al. 2021; *See* Selway River and Lochsa River chapters in this report). As discussed above, this is likely due to the warmer temperatures and higher productivity in the SFCR compared to the Lochsa River and Selway River systems.

MOUNTAIN WHITEFISH

A significant declining trend (2000 - 2018) in the mean density of MWF in the SFCR was detected. Declines in MWF abundance have been observed in other northern Idaho rivers, including the Lochsa River and Selway River (*See* Lochsa River and Selway River chapters in this report). Long-term declines in MWF have been documented in other populations across the southern portion of their range as well, including the Big Lost River and Kootenai River in Idaho, the Yampa River in Colorado, and the Madison River in Montana (Paragamian 2002; IDFG 2007; Boyer 2016). These surveys concluded that a variety of factors were likely responsible for the declines in MWF abundance including low flows, higher water temperatures, habitat alteration, irrigation diversions, nonnative fish interactions, disease, and harvest (IDFG 2007; Brinkman et al. 2013; Boyer 2016). The SFCR has a long history of habitat alteration and agricultural use which may have had historic impacts on local fish populations. However, agricultural use is primarily in the lower stream section which did not show a declining trend. While illegal harvest of WCT may be an issue, we believe harvest of MWF is low throughout the Clearwater River drainage. Thus, environmental factors may be more directly influencing the declines in density.

In the SFCR, environmental factors such as higher summer water temperatures and winter conditions are more likely contributing to the recent declining trends in MWF density. Increased water temperatures could affect the number of MWF observed through increased mortality (Jager et al. 1999; Copeland and Meyer 2011; Kennedy and Meyer 2015). Additionally, MWF eggs and fry have been shown to have a lower thermal tolerance than other salmonids such as RBT and WCT (Rajogopal 1979; Eaton and Scheller 1996; Brinkman et al. 2013). Mean monthly summer air temperatures in north-central Idaho have been above normal every year except one since 1996 (NOAA 2021). Additionally, water temperatures during our surveys have averaged > 20°C since 2010, compared to 16°C prior. As such, higher water temperatures could be impacting MWF movement, survival, and recruitment. Additionally, severe outbreaks of Proliferative Kidney Disease (PKD) have been observed in Montana, often occurring when the fish are stressed from heat (Hutchins et al. 2021). This disease is known to be present in Idaho (Phillips 2016; Hutchins et al. 2021). While no major fish kills have been directly observed, minor die-offs have been observed in Idaho rivers during summer months. Thus, PKD could also impact populations through lower level mortality.

In addition to warmer summer temperatures, climate change may cause more severe winter conditions through an increase in prevalence of anchor ice. This occurs when warmer temperatures reduce snow cover, which insulates against anchor ice formation (Butler 1979). Anchor ice, which forms on the bottom of river beds, can directly affect fish populations through direct mortality (increased stress, stranding, etc.) as well as impacts to redds and benthic invertebrate communities that serve as food sources (Butler 1979; Jakober et al. 1998; Brown et al. 2011). Anchor ice is known to be prevalent in SFCR tributaries (especially Red River), and lower snowfall could increase its formation and duration, and therefore its potential impacts on fish and habitat.

In contrast to overall density, no significant trend in density of MWF > 305 mm was calculated. This suggests that the trends in MWF populations observed in the SFCR are size-dependent, with declines occurring at smaller sizes. However, the high variability in density and few years of data makes it difficult to draw conclusions at this time. Within the SFCR, fewer juvenile MWF have been observed during snorkel surveys over the last 10 years compared to historic surveys (*IDFG unpublished data*; Roth et al. 2018*; Scott Putnam, personal communication*). Fish populations are often limited by recruitment, and changes in juvenile survival would have long-lasting effects on the population (Bradford and Cabana 1997; Pope et al. 2010). If changes in habitat or temperature regimes are occurring, a decline in juvenile abundance may be an early indicator, and would explain why we are seeing declining trends in the overall population.

Additional surveys will allow for a more thorough analysis of trends in MWF populations in the main-stem SFCR. However, the apparent downward trend in MWF density across the Clearwater River drainage and other parts of their historic range warrants a more detailed analysis.

SMALLMOUTH BASS

Smallmouth Bass were not observed in the main-stem SFCR sampling until 2014. Since then, they have been observed in low abundance. They have also been recently observed in the lower reaches of the Lochsa River and North Fork Clearwater River (*See* Lochsa River chapter in this report; Hand et al. 2021). Based on the size of SMB observed in 2018 (150 - 300 mm), it appears that reproduction is occurring the lower reaches of the SFCR. Smallmouth Bass colonization of salmonid spawning and rearing habitat has been documented throughout the Columbia River Basin (Lawrence et al. 2014; Rubenson and Olden 2017). Potential increases in SMB distribution is of concern, as these non-native fish can be a substantial predator of salmonids (Tabor et al. 1993; Naughton et al. 2004; Tiffan et al. 2020). Future surveys in the SFCR should continue to record observations of SMB, as they may experience a climate change related spread throughout the Clearwater River system (Rahel and Olden 2008).

MANAGEMENT RECOMMENDATIONS

1. Continue to evaluate trends in abundance and the size structure of Westslope Cutthroat Trout, Rainbow Trout, and Mountain Whitefish in the South Fork Clearwater River on a two year on, two year off basis.

2. Attempt to determine if RBT have a clipped adipose fin during snorkel surveys.

3. Continue to monitor Smallmouth Bass distribution and abundance in the South Fork Clearwater River to assess whether upstream colonization is increasing.

4. Support in-stream habitat improvement projects to increase densities of native fish in the South Fork Clearwater River.

5. Add signage along the South Fork Clearwater River alerting anglers to what the fishing regulations are for Westslope Cutthroat Trout.

Table 21. Fishes counted in each transect snorkeled in the main-stem South Fork Clearwater River, Idaho, in 2018.

					Number of fish						
River section	Transect name	Temp °C	Visibility (m)	Area (m^2)	Rainbow Trout	Chinook Salmon	Westslope Cutthroat Trout	Mountain Whitefish	Bull Trout	Brook Trout	Smallmouth Bass
Lower	8.5 KM	24.0	1.6	3,200	0	1	0	0	0	0	0
	13.4 KM	24.5	2.7	2,480	16	2	0	0	0	0	0
	18.2 KM	24.5	2.2	3,007	0	1	0	14	0	0	4
	23.0 KM	24.0	2.2	2,774	0	0	0	18	0	0	0
	28.5 KM	22.0	1.8	1,245	2	0	0	1	0	0	0
Middle	33.7 KM	22.0	1.7	1,215	0	1	0	4	0	0	0
	38.5 KM	23.0	1.3	1,634	13	1	0	9	0	0	0
	43.9 KM	21.0	1.4	3,213	5	3	0	13	0	0	0
	48.7 KM	21.0	1.4	3,243	3	0	2	27	0	0	0
	53.0 KM	20.0	1.4	2,445	21	0	0	43	0	0	0
	58.2 KM	18.0	1.6	1,980	0	5	1	3	0	0	0
	63.7 KM	18.5	1.7	1,843	40	1	1	3	0	0	0
	68.6 KM	18.0	1.3	408	7	0	0	0	0	0	0
	73.7 KM	20.0	2.0	645	21	0	0	1	0	0	0
Upper	78.3 KM	20.0	2.4	1,020	15	0	1	0	0	0	0
	83.9 KM	18.0	1.7	2,000	0	5	0	6	0	0	0
	88.7 KM	18.0	1.1	512	0	0	0	5	0	0	0
	93.9 KM	16.0	1.4	836	0	0	0	0	0	0	0
	98.7 KM	18.0	---	768	1	1	0	0	0	0	0
	103.2 KM	18.0	2.0	1,952	1	1	4	0	0	0	0
	Mean				7.3	1.1	0.5	7.4	0.0	0.0	0.2
	90% CI				0.3	0.0	0.0	0.2	---	---	0.0

Table 22. Densities (fish/100 m^2) of Westslope Cutthroat Trout observed during snorkel surveys of the South Fork Clearwater River, Idaho, from 2000 to 2018.

River section	Transect	2000	2010	2014	2017	2018
Lower	3.8 KM	0.00	0.00	0.00	0.00	0.00
	8.5 KM	0.00	0.00	0.00	0.00	0.00
	13.4 KM	0.00	0.00	0.00	0.00	0.00
	18.2 KM	0.07	0.00	0.00	0.04	0.00
	23.0 KM	0.00	0.00	0.00	0.00	0.00
	28.5 KM	0.00	0.00	0.00	0.00	0.00
	Mean	0.01	0.00	0.00	0.01	0.00
Middle	33.7 KM	0.00	0.00	0.06	0.00	0.00
	38.5 KM	0.44	0.00	0.00	0.00	0.00
	43.9 KM	0.00	0.00	0.00	0.00	0.00
	48.7 KM	0.05	0.07	0.00	0.14	0.06
	53.0 KM	0.00	0.08	0.00	0.00	0.00
	58.2 KM	0.14	0.15	0.23	0.06	0.05
	63.7 KM	0.00	0.41	0.29	0.00	0.05
	68.6 KM	0.00	0.00	0.00	0.00	0.00
	73.7 KM	0.00	0.33	0.16	0.00	0.00
	Mean	0.07	0.12	0.08	0.02	0.02
Upper	78.3 KM	0.10	0.12	0.00	0.00	0.10
	83.9 KM	0.08	0.48	0.10	0.22	0.00
	88.7 KM	0.00	0.63	0.00	0.00	0.00
	93.9 KM	0.00	0.18	0.12	0.00	0.00
	98.7 KM	0.13	0.32	0.00	0.00	0.00
	103.2 KM	0.04	0.14	0.00	0.00	0.20
	Mean	0.06	0.31	0.04	0.04	0.05
	Mean	0.05	0.14	0.05	0.02	0.02
	90% CI	0.04	0.07	0.03	0.02	0.02
	Mean > 305 mm	0.001	0.020	0.001	0.004	0.003
	90% CI	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01

Table 23. Intrinsic rate of change (r_{intr}) in density (fish/100 m^2) for Westslope Cutthroat Trout, Rainbow Trout, and Mountain Whitefish for snorkel surveys conducted in the South Fork Clearwater River, Idaho, from 2000 to 2018. Significance was set at α = 0.10.

	r_{intr}	90% CI	
Species	estimate	lower	upper
Westslope Cutthroat Trout			
all sizes	-0.052	-0.171	0.068
> 305 mm	0.083	-0.151	0.318
Rainbow Trout	-0.063	-0.153	0.028
Mountain Whitefish			
all sizes	-0.086	-0.149	-0.022
> 305 mm	-0.047	-0.132	0.037

Table 24. Intrinsic rate of change (r_{intr}) in density (fish/100 m^2) for Westslope Cutthroat Trout, Rainbow Trout, and Mountain Whitefish by river section for snorkel surveys conducted in the South Fork Clearwater River, Idaho, from 2000 to 2018. Significance was set at $\alpha = 0.10$.

Species	r_{intr} estimate	90% CI lower	90% CI upper
Westslope Cutthroat Trout			
Lower	-0.062	-0.166	0.042
Middle	-0.069	-0.189	0.051
Upper	-0.035	-0.194	0.125
Rainbow Trout			
Lower	-0.063	-0.207	0.081
Middle	-0.048	-0.165	0.070
Upper	-0.110	-0.194	-0.026
Mountain Whitefish			
Lower	-0.076	-0.283	0.130
Middle	-0.076	-0.129	-0.024
Upper	-0.115	-0.156	-0.074

Table 25. Minimum, maximum, and mean total length of Westslope Cutthroat Trout observed during snorkel surveys of the South Fork Clearwater River, Idaho, from 2000 to 2018.

		Length (mm)		
Year	*n*	Min	Max	Mean
2000	35	76	254	135
2010	49	76	432	214
2014	8	127	305	270
2017	9	76	406	186
2018	9	178	432	279

Table 26. Densities (fish/100 m^2) of Rainbow Trout observed during snorkel surveys of the South Fork Clearwater River, Idaho, from 2000 to 2018.

River section	Transect	2000	2010	2014	2017	2018
Lower	3.8 KM	0.00	0.00	0.58	0.05	0.00
	8.5 KM	0.00	0.00	0.00	0.00	0.00
	13.4 KM	0.00	0.07	0.00	0.24	0.65
	18.2 KM	1.34	0.00	0.00	0.00	0.00
	23.0 KM	0.13	0.17	0.24	0.00	0.00
	28.5 KM	1.57	0.00	0.10	0.48	0.16
	Mean	0.51	0.04	0.15	0.13	0.13
Middle	33.7 KM	0.00	0.04	6.68	1.45	0.00
	38.5 KM	8.32	3.38	5.33	3.87	0.80
	43.9 KM	0.74	0.26	1.07	0.33	0.16
	48.7 KM	5.09	0.34	0.54	0.85	0.09
	53.0 KM	1.52	1.07	0.63	0.35	0.86
	58.2 KM	3.54	0.45	5.05	3.20	0.00
	63.7 KM	5.19	1.56	5.17	5.17	2.17
	68.6 KM	8.43	0.63	6.41	1.59	1.72
	73.7 KM	6.14	0.99	5.02	2.66	3.26
	Mean	4.33	0.97	3.99	2.16	1.00
Upper	78.3 KM	8.09	1.22	5.44	2.69	1.47
	83.9 KM	2.57	0.96	0.70	2.47	0.00
	88.7 KM	3.32	1.96	0.38	0.34	0.00
	93.9 KM	2.13	1.14	0.00	0.36	0.00
	98.7 KM	3.31	1.34	0.00	0.00	0.13
	103.2 KM	2.02	2.03	0.00	0.28	0.05
	Mean	3.57	1.44	1.09	1.02	0.28
	Mean	3.02	0.84	2.06	1.26	0.55
	90% CI	2.77	0.86	2.54	1.47	0.33

Table 27. Minimum, maximum, and mean total length of Rainbow Trout observed during snorkel surveys of the South Fork Clearwater River, Idaho, from 2000 to 2018.

		Length (mm)		
Year	*n*	Min	Max	Mean
2000	997	75	381	125
2010	283	50	330	157
2014	542	50	356	99
2017	348	50	356	92
2018	145	50	330	153

Table 28. Densities (fish/100 m^2) of Mountain Whitefish observed during snorkel surveys of the South Fork Clearwater River, Idaho, from 2000 to 2018.

River section	Transect	2000	2010	2014	2017	2018
Lower	3.8 KM	0.00	0.00	0.58	0.00	0.00
	8.5 KM	0.09	0.07	0.00	0.00	0.00
	13.4 KM	0.19	0.00	0.11	0.00	0.00
	18.2 KM	0.61	0.17	1.82	0.31	0.47
	23.0 KM	2.90	0.00	1.04	0.31	0.65
	28.5 KM	1.80	0.06	0.68	0.00	0.08
	Mean	0.93	0.05	0.70	0.10	0.20
Middle	33.7 KM	0.40	0.26	0.06	0.14	0.33
	38.5 KM	0.81	4.17	0.27	0.27	0.55
	43.9 KM	1.04	0.09	0.45	0.33	0.40
	48.7 KM	1.95	0.27	1.21	0.85	0.83
	53.0 KM	1.58	0.15	1.26	0.35	1.76
	58.2 KM	1.07	0.25	0.00	0.19	0.15
	63.7 KM	4.60	0.14	0.44	0.16	0.16
	68.6 KM	0.70	0.21	1.28	0.40	0.00
	73.7 KM	2.52	0.00	1.88	0.00	0.16
	Mean	1.63	0.62	0.76	0.30	0.48
Upper	78.3 KM	1.55	0.49	0.45	0.10	0.00
	83.9 KM	1.41	0.17	0.91	0.44	0.30
	88.7 KM	1.59	0.25	0.96	0.00	0.98
	93.9 KM	1.68	0.42	0.12	0.12	0.00
	98.7 KM	1.03	0.24	0.00	0.00	0.00
	103.2 KM	1.62	2.17	0.09	0.37	0.00
	Mean	1.48	0.62	0.42	0.17	0.21
	Mean	1.38	0.46	0.65	0.21	0.32
	90% CI	0.38	0.35	0.21	0.08	0.16
	Mean > 305 mm	0.39	0.15	0.24	0.09	0.24
	90% CI	0.27	0.10	0.20	0.05	0.18

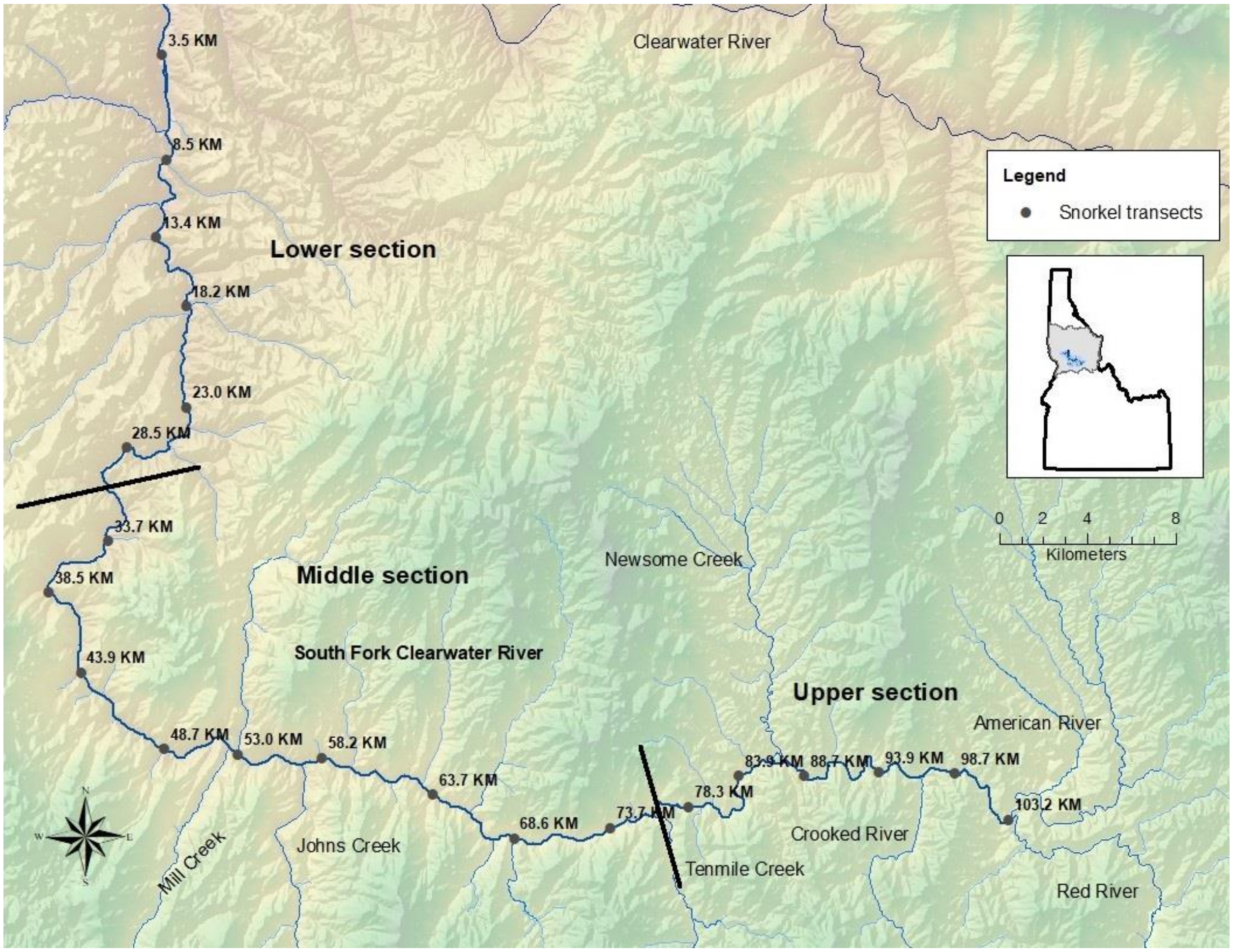

Figure 51. Map showing locations of snorkel transects surveyed on the South Fork Clearwater River, Idaho, in 2018.

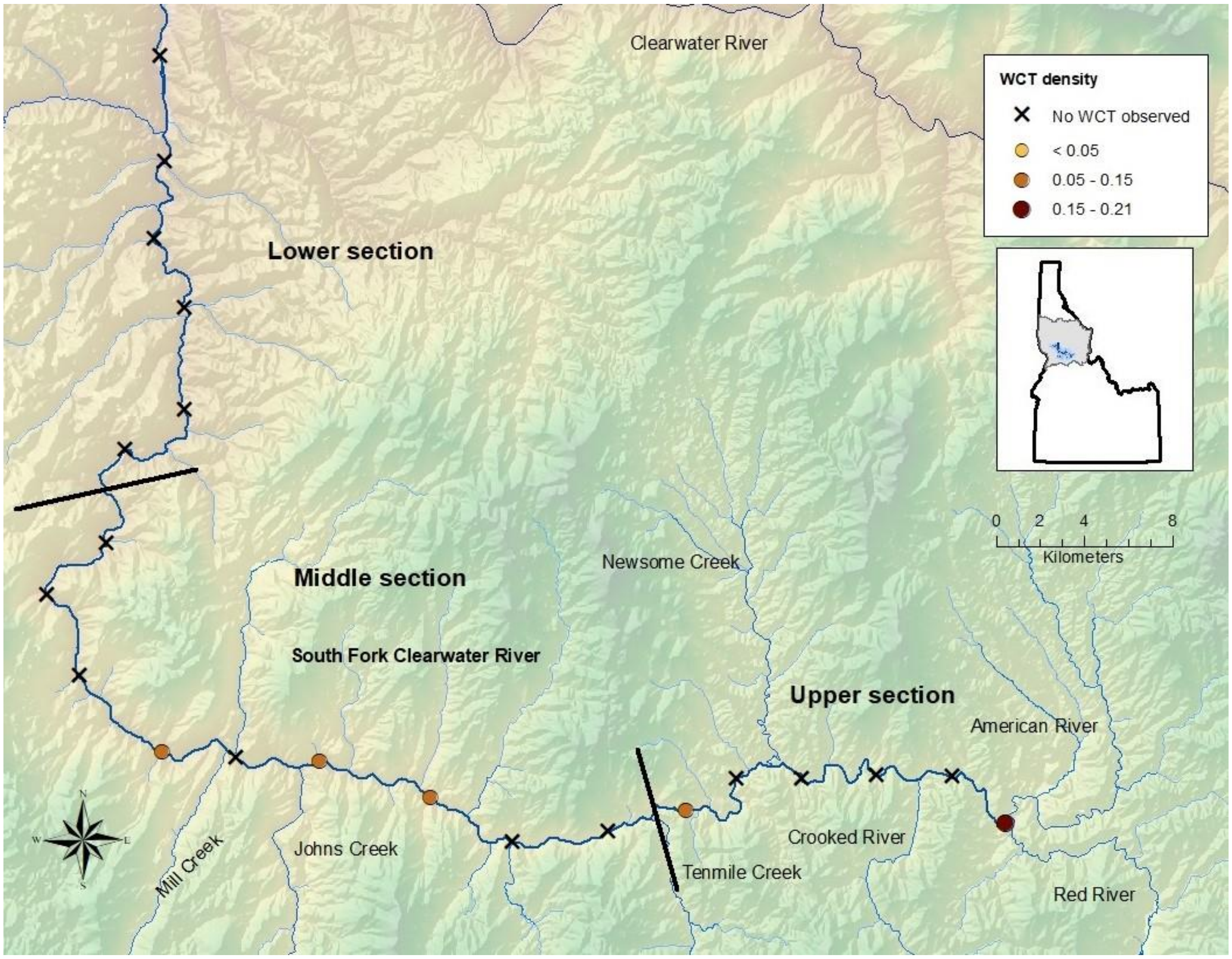

Figure 52. Densities (fish/100 m^2) of Westslope Cutthroat Trout (WCT) observed in each snorkel transect surveyed in the South Fork Clearwater River, Idaho, in 2018.

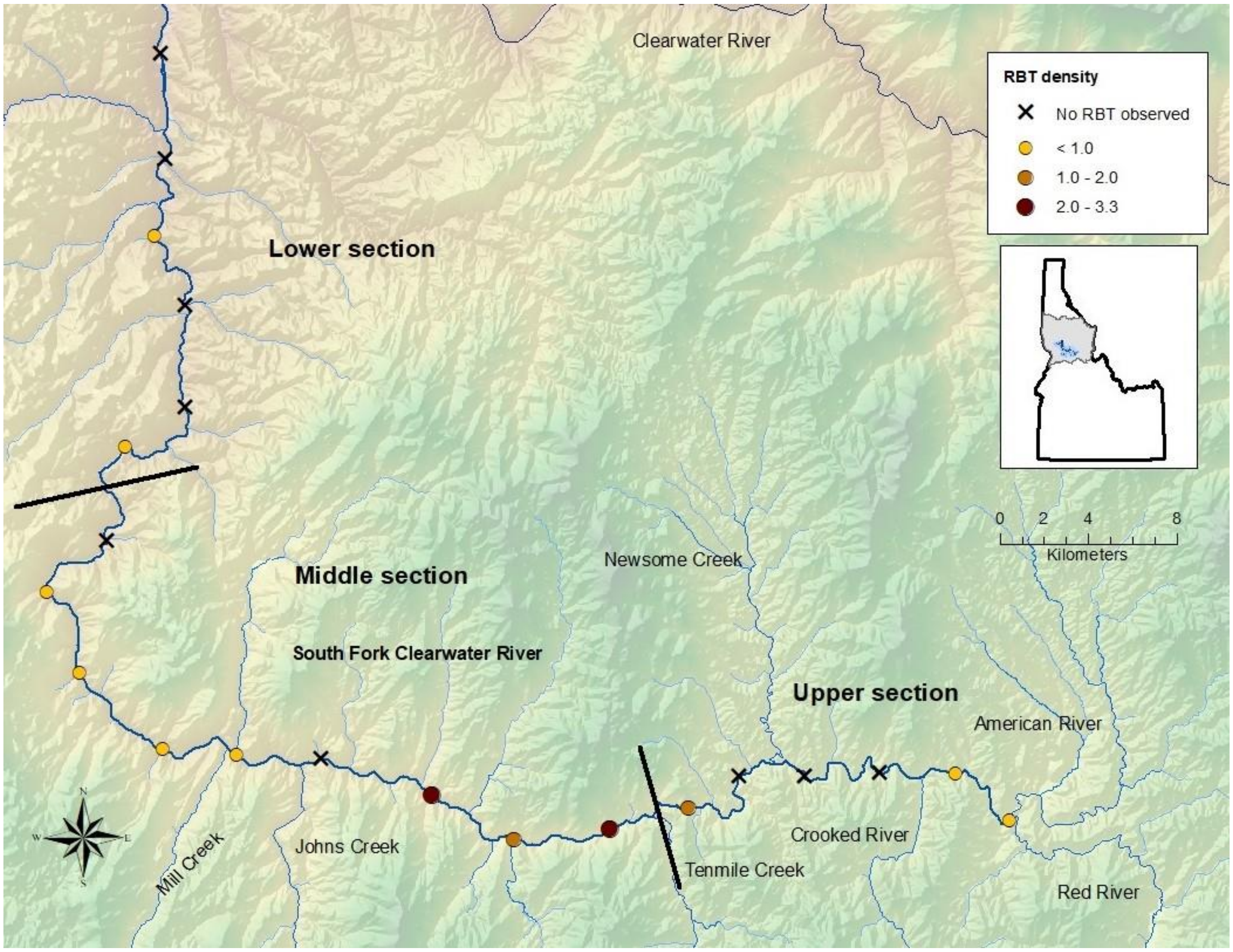

Figure 53. Density (fish/100 m^2) of Rainbow Trout observed in each snorkel transect surveyed in the South Fork Clearwater River, Idaho, in 2018

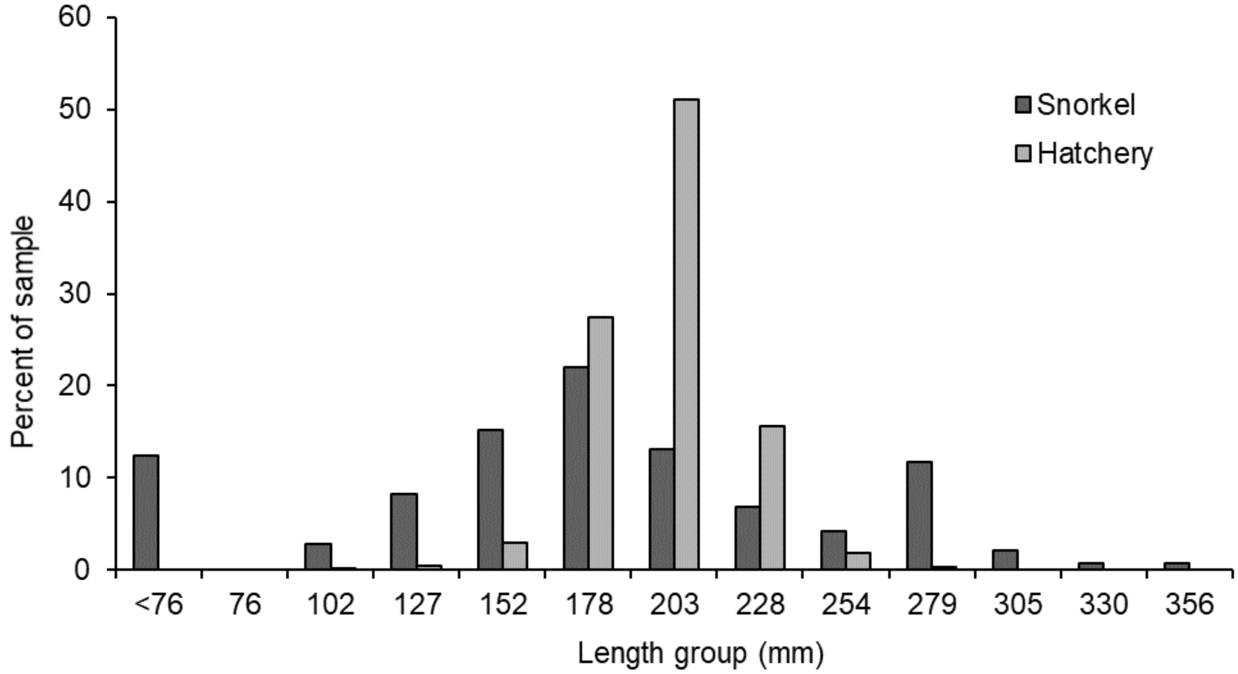

Figure 54. Length-frequency distributions of Rainbow Trout observed in snorkel transects (Snorkel; August) and steelhead smolts stocked (Hatchery; March) in the South Fork Clearwater River, Idaho, in 2018.

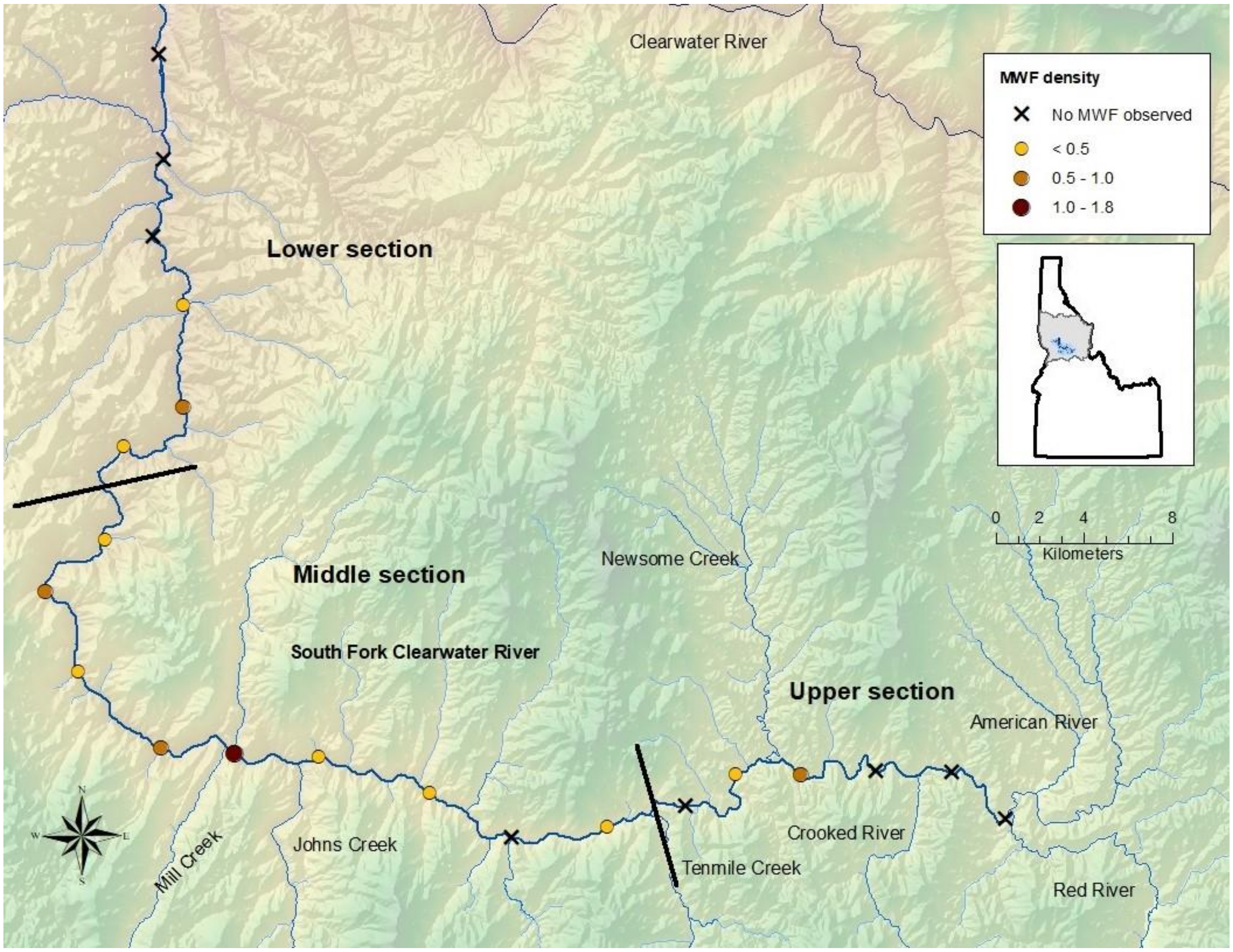

Figure 55. Densities (fish/100 m^2) of Mountain Whitefish (MWF) observed in each snorkel transect surveyed in the South Fork Clearwater River, Idaho, in 2018.

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MOUNTAIN LAKES MONITORING IN CONSIDERATION OF AMPHIBIAN RISK ASSESSMENT IN NORTH CENTRAL IDAHO

ABSTRACT

We conducted the 12th year of a 20-year study evaluating whether current fisheries management strategies in high mountain lakes of North Central Idaho adequately balance recreational fishing opportunity and provide for the long-term persistence of amphibian populations. Preliminary analysis suggests gill net CPUE for fish has declined over time in Middle Wind, Hungry, and Siah lakes; however, there was no statistically significant trend in CPUE across all project lakes containing fish. There was no significant trend in Columbia Spotted Frog (CSF) or Long Toed Salamander (LTS) presence in high mountain lakes surveyed in 2017. Long-toed salamander presence was negatively correlated with the habitat variable "fish presence", while CSF presence was not correlated with any habitat variables. Both species were positively correlated with the temporal variable "Julian Date^2". For surveys conducted from 2014 to 2017, the composite detection probability for all life stages detected during visual encounter surveys was 0.92 for CSF and 0.62 for LTS. Declining gill net CPUE could result from annual variation in weather at the time of the survey, or broader drivers such as reduced food resources or recruitment failure caused by severe winter or summer conditions. Long-term trends in CSF and LTS presence were consistent with previous findings and indicate that these populations have remained stable throughout the duration of this study. The negative relationship between LTS and fish presence aligns with previous findings in this study, and other studies conducted throughout their range. Based on the preliminary trends in amphibian populations with our study lakes, current fisheries management of high mountain lakes appears adequate for balancing fishing opportunity with the long-term persistence of amphibians. However, a more detailed analysis at the end of this study will be necessary to determine impacts on a larger scale, and especially in HUCs that include lakes currently within the IDFG high mountain lake stocking program.

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INTRODUCTION

Amphibian population reduction and species extinction has given urgency to amphibian conservation, inventory efforts to determine baseline data, and monitoring to determine trends in amphibian populations (Houlahan et al. 2000; Stuart et al. 2004; Beebee and Griffiths 2005; Orizaola and Brana 2006). Potential factors in amphibian population decline are numerous and include: habitat modification/fragmentation, introduction of predators/competitors, increased UV-B radiation, changes in precipitation/snowpack, and pathogen infection (Alford and Richards 1999; Corn 2000; Marsh and Trenham 2001; Pilliod and Peterson 2001). Throughout the North Central Mountains of Idaho, direct (predation) and indirect (resource competition, habitat exclusion, and population fragmentation) impacts on amphibian populations from introductions of trout into historically fishless lakes are also a cause for concern (Semlitsch 1988; Figiel and Semlitsch 1990; Bradford et al. 1993; Brönmark and Edenhamn 1994). Trout have been stocked into high mountain lakes to provide recreational opportunities to backcountry visitors. As much as 95% of previously and/or currently stocked high mountain lakes throughout the western United States that were once fishless, now contain fish through regular stocking efforts or self-sustaining populations from legacy stocking efforts (Bahls 1992). It is estimated that 96% of lakes within the Nez Perce-Clearwater National Forest were historically fishless as the headwater area topography where lakes are located is relatively steep (Murphy 2002). According to historical stocking records, some lakes in North Central Idaho were stocked as early as the 1930s (Murphy 2002). Out of the estimated 3,000 mountain lakes in Idaho, approximately 1,355 lakes (45%) are stocked or have self-sustaining fish populations (IDFG 2013).

Mountain lake ecosystems in North Central Idaho contain amphibians such as long-toed salamander *Ambystoma macrodactylum* (LTS) and Columbia spotted frog *Rana luteiventris* (CSF), although Idaho giant salamander *Dicamptodon aterrimus*, western toad *Bufo boreas*, and Rocky Mountain tailed frog *Ascaphus montanus* may also be present. Common reptiles found at these mountain lakes may also include common Garter snake *Thamnophis sirtalis* and western terrestrial Garter snake *T. elegans*, both of which were historically (before fish introductions) the main amphibian predators (Murphy 2002).

Surveys have found that CSF occurrence (and breeding occurrence) in the Clearwater Region was not significantly different in lakes with or without fish after accounting for habitat effects (CSF were positively associated with increasing amounts of sedge meadow perimeter and silt/organic substrate) (Murphy 2002). However, CSF abundance at all life stages was significantly lower in lakes with fish than without fish (Murphy 2002). In contrast, LTS larvae and/or breeding adult occurrence and abundance (adults are typically terrestrial except to breed) was significantly less common in lakes with fish then lakes without fish (Murphy 2002). However, where native (not stocked) Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* (WCT) existed in lakes, the impact on LTS was not as severe as compared to lakes that were historically fishless and later stocked with trout (Murphy 2002). Other studies have examined relationships between introduced trout and salamanders. Direct negative impacts by fish on amphibian populations have been mostly attributed to trout preying upon amphibians when they are at a larval stage, although trout may also cause salamanders to avoid lakes previously used as breeding sites (Kats et al. 1993; Figiel and Semlitsch 1990; Bradford et al. 1993; Knapp 1996; Pilliod et al. 1996; Graham and Powell 1999; Murphy 2002).

Introduced fish populations may also indirectly impact amphibian gene flow, recolonization, and subsequent persistence. The degree of gene flow in mountain lake amphibians likely relies on connectivity between higher and lower elevations subpopulations (with low gene flow). Gene flow may also occur between neighboring lakes that are not necessarily within the same wet stream migration corridor when overland dispersal is not drastically limited by headwater topography, precipitation, and or canopy cover (Murphy 2002). Long-toed salamander within north-central Idaho are panmictic (randomly interbreeding populations), with high levels of within population variation providing evidence that populations are not evolving in complete isolation (Tallmon et al. 2000). Amphibian populations or demes in these headwater areas likely never evolved with native fish and may lack the appropriate defensive, behavioral, or chemical responses to coexist with introduced fish populations (Kats et al. 1988).

Westslope Cutthroat Trout, Rainbow Trout *O. mykiss* (RBT), RBT x WCT hybrids, and Brook Trout *Salvelinus fontinalis* (BKT) are the most common introduced fish species in high mountain lakes in the Clearwater Region. Although, many lakes within the study area have a stocking history that may include Yellowstone Cutthroat Trout *O. bouvieri*, California Golden Trout *O. m. aguabonita* (last stocked in 1990 in the Clearwater Region - Steep Lakes), Arctic Grayling *Thymallus arcticus* (last stocked in 1982 in the Clearwater Region - Bald Mountain Lake), and various forms of trout hybrids. The term “introduced western trout” may be more appropriate for *Oncorhynchus* species in these lakes where natural reproduction is occurring, as the degree of hybridization is unknown in lakes where multiple species have been stocked (Behnke 1992). The Clearwater Region currently stocks 87 of its 711 high mountain lakes. Most lakes are stocked with fingerling WCT on a three-year rotation by fixed-wing aircraft.

Certain species of introduced trout tend to have a greater impact on amphibian occupancy than others (Murphy 2002). Brook Trout tend to impact CSF and especially LTS occurrence and breeding to a greater extent than the presence of WCT or RBT. This impact is derived from differences in fish spawning times/behavior and variations in amphibian habitat usage just after ice off conditions in mountain lakes (Murphy 2002). Westslope Cutthroat Trout and RBT in these lakes spawn in spring/summer which often coincides with times that amphibian breeding occurs. As a result, both fish species are typically preoccupied with spawning in inlets or outlets while amphibians are typically breeding within the lake itself. This difference in spawning habitat use may allow amphibians to breed with fewer disturbances by WCT and RBT (Murphy 2002). In contrast, BKT are fall spawners and are actively moving and foraging throughout the lake in spring and are more likely to prey upon any amphibian life stage and/or harass breeding adults (Murphy 2002). Furthermore, BKT tend to be more benthic oriented (where salamanders usually occur), utilize larger prey items, and attain higher densities within mountain lakes than *Oncorhynchus* species (Griffith 1974). Columbia spotted frog do not tend to be impacted by BKT presence to the same magnitude as LTS because of their different habitat associations and shorter length of larval stages (Griffith 1974; Bahls 1992; Murphy 2002).

Long-toed salamander occupies a wide range over the western United States and Canada. The majority of LTS in Idaho sub-alpine lakes have a two year larval stage, making them susceptible to predation by fish for a longer period of time. Studies suggest that they are more susceptible to impacts by introduced fish than the CSF (Murphy 2002). Conclusive evidence of LTS decline is insufficient (Graham and Powell 1999). For this reason, a long-term monitoring project (20 years) was initiated in the Clearwater Region to provide knowledge of the amphibian population dynamics within the north-central mountains of Idaho. Long-term monitoring of mountain lakes will allow for amphibian population trends to be identified and will give managers the ability to determine whether sufficient fishless habitat exists to support amphibian populations into the future.

OBJECTIVES

1. Evaluate whether high mountain lakes within the IDFG Clearwater Region that have fish are less likely to have amphibians.

2. Assess whether current fisheries management strategies in high mountain lakes of North Central Idaho adequately balance recreational fishing opportunity and provide for the long-term persistence of amphibian populations.

STUDY AREA

The 74 lakes selected for this study are located within the Nez Perce-Clearwater National Forests, located in north-central Idaho (Figure 56). High mountain lakes within this study are primarily located in wilderness areas (Selway-Bitterroot, Gospel Hump, and Frank Church River of No Return Wilderness) within the Nez Perce-Clearwater National Forest, with two lakes located outside of wilderness boundaries. There are nine HUC5 creek drainages in the study: Goat, Upper Meadow, Big Harrington, North Fork Moose, Storm, Running, Warm Springs, Old Man, and Bargamin creeks. Hiking and multi-day backpacking trips are required to access all of the high mountain lakes in this study.

METHODS

The study design and protocol for this long-term evaluation utilized an amphibian risk assessment model developed through previous studies and inventories of mountain lakes conducted within north-central Idaho. This model is based on the amount of fishless habitat that exists within a watershed at the HUC5 level. At the individual HUC5 watershed level, it is assumed monitoring will be able to examine conditions that may dictate local response in the interactions of stocked fish and native amphibian populations to provide a more defined opportunity for prioritized management action (Murphy 2002). While there are many risk factors associated with amphibian declines, our assessment focused on considering impacts that may be associated with native and stocked fish in lakes on a HUC5 watershed basis. The amphibian risk assessment model for these high mountain lake ecosystems has four categories: control (no risk), low, moderate, and elevated.

- *Control or no risk* – watershed has never experienced fish introductions through stocking activities.
- *Low* – At least 50% of the lakes within a watershed are fishless <u>AND</u> a minimum 20% of the lake surface area within the watershed is fishless.
- *Moderate* – 50% of lakes within a watershed are fishless <u>OR</u> 20% of surface area is fishless.
- *Elevated* – Meets neither requirement, less than 50% of the lakes within a watershed are fishless <u>AND</u> less than 20% of the surface area within the watershed is considered fishless.

Two HUC5 watersheds were selected randomly from each of the amphibian risk categories (region-wide from all HUC5 watersheds that contained lakes) for sampling. This resulted in eight HUC5 watersheds containing 72 lakes within the Nez Perce-Clearwater National Forest. In 2013, a third randomly selected HUC5 watershed (Big Harrington Creek) was added to increase the sample size of fishless control lakes, bringing the study's total to nine watersheds that contain 74 lakes (Hand et al. 2016). Attempts will be made to sample all lakes within a selected HUC5 watershed within the same field season. The 20-year period for the high mountain lakes long-term monitoring project will allow for each of these lakes be sampled ~6 times each. The repetition of sampling events will allow for comparisons to be made within (for trends) and between watersheds (for comparisons among amphibian risk classes). In addition, repetition of sampling events will address the normal patterns of recruitment fluctuations often common among amphibian populations. Sampling frequency and rotation order are adjusted as needed due to weather, trail, and fire conditions.

FIELD SAMPLING

In 2018, 20 lakes were selected to be surveyed within the Big Harrington Creek, Goat Creek, North Fork Moose Creek, Old Man Creek, Running Creek, and Upper Meadow Creek HUC5 units (Figure 56). Field sampling was conducted following the protocol used throughout the duration of this project, and revised after the 2013 field season to improve the accuracy and comparability of results from year-to-year (Hand et al. 2016). Beginning with the 2014 field season we introduced the use of multiple visual encounter surveys (VES) at a lake within a 24-hour timeframe to increase the probability of detection for amphibians, and to allow for estimating detection probabilities. These surveys were separated by at least three hours and conducted during different parts of the day (i.e. morning and afternoon), or even different days, when time allowed.

DATA ANALYSIS

Catch-per-unit-effort

We evaluated trends in gill net CPUE (fish/h) across all project lakes containing fish using least squares regression with survey number (e.g. 1st survey conducted, 2nd survey, etc.) as the independent variable and $\log_e$ transformed CPUE as dependent variables (Maxell 1999; Kennedy and Meyer 2015). We did not evaluate trends in CPUE based on year as lakes are not surveyed in the same year, or at consistent intervals. The rate of change (r_{intr}) in CPUE is determined by the slope of the regression line fit to these data. A 90% CI was calculated for r_{intr} to determine significance, where the trend is considered significant when $r_{intr} \neq 0$ and the error bounds do not include 0. We used a significance level of $\alpha = 0.10$. We also assessed relationships between mean gill net CPUE by species and elevation for all project lakes.

Detection probabilities

We evaluated detection probabilities of CSF and LTS by comparing data collected from 1988 to 2018. Statistical analysis followed the methods developed in Hand et al. (2016) and Hand et al. (2018). Detection probabilities considered any zero count as a “failed detection”. Composite scores were developed for both CSF and LTS in order to consider both amphibian species across life stages of adult, sub adult, and larvae. Logarithmic transformation was used for larvae abundance to better fit these data with adult and sub-adult data. Detection probabilities were used as a correction factor for the percentage of lakes with CSF and LTS present in a survey year.

Amphibian presence

We used Generalized Linear Mixed Effect Models (GLMs) to evaluate long-term trends in CSF and LTS presence, and the effect of habitat variables (Fish Presence, Elevation, Maximum Depth, Percent Emergent Vegetation, Percent Fines, Percent Littoral Zone, Julian Date, and Julian Date^2) on the occurrence of CSF and LTS. The GLMs used to evaluate the effect of habitat variables on CSF and LTS presence were built hierarchically in a stepwise fashion. These analyses utilized the 63 project lakes for which we have both historic data (collected prior to this study) and three rounds of surveys within this study (collected from 1988 to 2018). Each model became progressively less and less complex (eliminating variables built into the model) in order to see which variables had significant impacts on amphibian presence. The GLMs followed statistical analysis source code procedures “Zuur_fxns” (Zuur et al. 2009) and code packages “gdata” (Warnes et al. 2017), “plyr” (Wickham 2011), and “lattice” (Sarkar 2008) which allowed building of data frames for data analysis. Statistical analysis for 2018 followed the methods outlined in Hand et al. (2016), and Hand et al. (2018). R Studio (R version-3.5.1 (2018-07-02) “Feather Spray”) was used to conduct the statistical analyses. Long-term trends in CSF and LTS presence were evaluated using models developed by Broström (2013). A full evaluation including all project lakes will be conducted at the end of this project.

Lake desiccation

To evaluate lake desiccation within our study lakes, we compared photos collected in 2018 to previous surveys conducted since 2006 and Google Earth imagery collected from 1998 to 2018. Changes in lake surface elevation were evaluated by comparing imagery across this time period.

RESULTS

FISH SURVEYS

Fish were present in 35% of the lakes surveyed in 2018 (Table 29). Gill net CPUE (fish/h) ranged from 0.1 to 5.4 fish/h (Table 30). For the lakes surveyed in 2018, gill net CPUE has been stable over the course of this study (Table 31; r_{intr} = -0.307; 90% CI bounds = -1.120, 0.507). This was consistent with findings in 2017.

COLUMBIA SPOTTED FROG

The detection probability for all life stages of CSF was 0.92 for multiple VES surveys. For all surveys conducted from 2014 to 2018, the detection probability for CSF was 0.92.

Columbia spotted frogs were observed in 75% (n = 15) of lakes surveyed (Table 29). Through 2018, 63 of the 74 project lakes now have historic data and three rounds of data from this project. For these 63 lakes, there was no trend in CSF presence (P = 0.213; Figure 57). Of the 15 lakes where CSF were observed, 44% contained fish.

Occurrence of all life stages of CSF (presence/absence) was not correlated with any habitat variables, including “Fish Presence” (Table 32). However, it was positively correlated with the temporal variable “Julian Date”, and negatively correlated with “Julian Date^2” (Table 32).

LONG-TOED SALAMANDER

The detection probability for all life stages of LTS was 0.62 for multiple VES surveys. For all surveys conducted from 2014 to 2018, the detection probability for LTS was 0.62.

Long-toed salamanders were observed in 44% of lakes surveyed (Table 29). Through 2018, 63 of the 74 project lakes now have both historic data and three rounds of data from this project. For these lakes, there was no trend in CSF presence (P = 0.781; Figure 57). Of the seven lakes where LTS were observed, none contained fish.

Occurrence of all life stages of LTS (presence/absence) was positively correlated with the habitat variable “Fish Presence” (Table 32). It was also positively correlated with the temporal variable “Julian Date”, but negatively correlated with “Julian Date^2” (Table 32).

LAKE DESSICATION

Substantial changes in surface elevation were observed in Upper and Lower Section 26 (Figure 58), Big Harrington #1 and #6 (Figure 59), and Section 27 (Figure 60) lakes. All of these except Section 27 Lake were completely dry at the time of our surveys. There were no amphibians present in or around the four dry lakes.

DISCUSSION

FISH SURVEYS

There was no statistically significant trend in gill net CPUE across all project lakes containing fish. This aligns with analysis conducted in 2017 (Hand et al. 2021). A concern was that our periodic surveying could suppress fish populations in lakes with low productivity, poor recruitment, and/or slow growth (Gray 2013). With declining CPUE occurring in only a few lakes surveyed in 2017, suppression from our sampling does not appear to be the cause. Changes in CPUE is likely a result of annual variation in weather at the time of the survey, or broader drivers such as reduced food resources or recruitment failure caused by severe winter or summer conditions (Armstrong and Knapp 2004; Parker et al. 2008). However, potential changes in fish populations are worth monitoring as we conduct additional surveys. A more detailed analysis of factors that may be influencing CPUE is beyond the scope of this report, but should be evaluated further if we continue to observe declining CPUEs in some lakes.

COLUMBIA SPOTTED FROG

The composite detection probability for all life stages of CSF was 0.92 for multiple VES surveys conducted from 2014 to 2018. Annual detection probabilities have stayed within a small range (0.89 to 0.95). This indicates that our sampling has remained consistent on an annual basis, and that detection probabilities are similar annually in spite of the potential for biases associated with differences in weather, lakes visited each year, and observers.

Our preliminary analysis indicates CSF presence has remained stables throughout the duration of this study. This was consistent with previous findings (Hand et al. 2021). At this time, the data suggest that any impacts on amphibian populations by fish have likely already occurred, and they have reached a point of stability and are not in a continued state of decline (Sexton and Phillips 1986; Knapp and Matthews 2000; Pilliod and Peterson 2001).

Columbia spotted frog presence was not correlated with any habitat variables, including “Fish Presence”. Habitat variables such as emergent vegetation and percent littoral zone have often been correlated with CSF presence and abundance, especially in lakes containing fish (Pilliod and Peterson 2001; Bull and Marx 2002). The lack of correlation in our study is likely due to the presence of CSF at 93% of study lakes, including 93% of lakes that contain fish. At every lake where fish were present there was concurrent presence of CSF. This is important to consider within the framework of this study, as it appears historic fish stocking is not having a continued impact on CSF populations.

In contrast, the temporal variable “Julian Date^2” was a significant predictor of CSF presence. Amphibian populations are known to change temporally due to environmental factors and differences in seasonal behavior (Bailey et al. 2004; Lohr and Haak 2011). Additionally, daily variation in detection rates can occur, as weather directly influences the efficacy of the VES. Amphibians may be less active, and therefore less noticeable, during colder weather conditions or storms (Lohr and Haak 2011). Therefore, yearly variation in weather, snow conditions, and even day-to-day conditions within a given high mountain lake will influence the observable amphibian populations. Given considerations of amphibian population age structure, weather, and individual surveyor bias, VES surveys should continue to be performed twice at each lake to improve likelihood of detection.

LONG-TOED SALAMANDER

The composite detection probability for all life stages of LTS was 0.62 for multiple VES surveys conducted from 2014 to 2018. Annual detection probabilities have stayed within small ranges (0.55 to 0.65). This indicates that our sampling has remained consistent on an annual basis, and that detection probabilities are similar annually in spite of the potential for biases associated with differences in weather, lakes visited each year, and observers.

Our preliminary analysis indicates LTS presence has remained stables throughout the duration of this study. This was consistent with previous findings (Hand et al. 2021). At this time, the data suggest that any impacts on amphibian populations by fish have likely already occurred, and they have reached a point of stability and are not in a continued state of decline (Sexton and Phillips 1986; Knapp and Matthews 2000; Pilliod and Peterson 2001).

Long-toed salamander presence was negatively correlated to fish presence. This relationship aligns with previous findings in this study, and other studies conducted throughout their range (Murphy 2002; Pearson and Goater 2009; Hand et al. 2016; Kenison et al. 2016). Since the beginning of this study, LTS have been observed at least once in 95% of all fishless lakes within this study, but only 48% of lakes containing fish. The negative impact of fish on LTS presence has been attributed to their two-year larval stage, where they have longer exposure to fish predation during their vulnerable aquatic rearing (Pilliod and Peterson 2001; Pearson and

Goater 2009). Greater overall predation on larval LTS relative to larval CSF likely explains why CSF can co-occur with fish in lakes where LTS are not detected or are detected only in low densities. Although LTS abundance and distribution may have stabilized, fish presence has a direct impact on the distribution of this species.

As with CSF, the temporal variable "Julian Date^2" was a significant predictor of LTS presence. Variation in detection related to Julian date has been explained by within-year variation in microclimate and localized growth conditions that cause amphibians to metamorphose at different times in different sub-basins (Pilliod and Peterson 2001). Another explanation for daily variation in detection rates is that weather directly influences the efficacy of the VES. Amphibians may be less active, and therefore less noticeable, during colder weather conditions or storms (Lohr and Haak 2011). Therefore, yearly variation in weather, snow conditions, and even day to day conditions within a given high mountain lake will influence the observable amphibian populations. Given considerations of amphibian population age structure, weather, and individual surveyor bias, VES surveys should continue to be performed twice at each lake to improve likelihood of detection.

LAKE DESICCATION

The issue of lake desiccation became more apparent in 2018, as five lakes had experienced large changes in surface elevation, with four being completely dry. Additionally, Eagle Creek Lake (not surveyed in 2018) has been found to be dry during recent surveys as well (Hand et al. 2019). Global climate change is a concern for amphibian populations in high mountain lakes, as it may magnify seasonal drying trends and lead to the permanent elimination of some water bodies and drastically change hydro periods in others (Gerick et al. 2014; Ryan et al. 2014). In fact, the introduction of nonnative trout and the loss of ephemeral habitats are considered to be two of the most significant anthropogenic challenges faced by montane wetland ecosystems (Ryan et al. 2014). Higher summer water temperatures coupled with shallower wetlands can increase amphibian mortality (Duarte et al. 2012; Gerick et al. 2014). The more significant the pond drying, the more significant the loss of amphibians will be compared to similar lakes that do not experience drying (Semlitsch and Wilbur 1988). In Yellowstone National Park, changes in hydrology from climate change are thought to be driving the desiccation of kettle ponds and wetland areas, which in turn have resulted in significant losses of Columbia spotted frog and other amphibians since 1992 (McMenamin et al. 2008). While many lakes are known to be ephemeral or intermittent, there appears to be a trend for smaller lakes in our project to fully desiccate. Google Earth imagery, USGS surveys, and our data all indicate that fluctuations in lake level and amphibian presence occur, though the magnitude of impact on amphibian populations is not known.

If lake desiccation is occurring with increased frequency, it could cause the loss of source populations of amphibians. Given that most of the lakes found to be dry are small, this could also be a loss of refuge areas where amphibians can survive without the predation of introduced trout. Populations of amphibians in high mountain lakes are not restricted to a single body of water, except in very isolated ponds (Funk et al. 2005). Instead, they are known to migrate from shallow breeding ponds to deep (>3m) lakes to overwinter (Pilliod et al. 2001). However, migration is reduced in more severe terrain and higher altitudes, which in turn reduces gene flow and genetic diversity of CSF and LTS (Funk et al. 2005; Pilliod et al. 2001). The loss of ephemeral ponds, small breeding ponds, overwintering lakes, and other montane wetland areas could have a tremendous impact on amphibian populations in the future (Ryan et al. 2014). More study on the amphibian populations of montane wetlands/ponds/lakes that are known to go dry would help to better understand the risk that climate change induced changes in hydrology could pose to the high mountain lake amphibian populations of the Clearwater Region, Idaho. While this preliminary analysis of lake desiccation suggests it could negatively impact amphibian populations, we recommend a more thorough analysis of this topic in the project completion report.

CONCLUSIONS

When comparing historic LTS and CSF inventories to survey data collected from 2006 to 2018, there have been no significant population trends amongst either species of amphibian. Therefore, when looking at these data collected over the entirety of this project, there has not been a notable decline in amphibian population across these lakes. This suggests that populations have remained consistent throughout the duration of the study, indicating that if amphibian populations were negatively impacted by fish presence, it likely occurred prior to this study, and have now stabilized. However, a more detailed analysis at the end of this study will be necessary to fully evaluate this stocking program to ensure it meets the needs of both anglers and amphibian conservation.

MANAGEMENT RECOMMENDATIONS

1. Continue evaluating whether high mountain lakes within the IDFG Clearwater Region that have fish are less likely to have amphibians.

2. Continue to assess whether current fisheries management strategies in high mountain lakes of North Central Idaho adequately balance recreational fishing opportunity and provide for the long-term persistence of amphibian populations.

3. Explore the utilization of eDNA collection in order to increase detection probabilities and to correlate amphibian presence and absence with eDNA data.

4. Reassess the amphibian risk assessment category system utilizing count data as well as presence and absence data.

5. Evaluate potential differences in impacts of individual salmonid species on amphibian presence and counts.

6. Investigate the nature of montane ephemeral ponds and desiccation trends of mountain lakes and their impacts on amphibian populations.

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Table 29. Presence and counts of Columbia spotted frog (CSF), long-toed salamander (LTS), and fish in high mountain lakes surveyed in the Clearwater Region, Idaho, in 2018.

Lake	Risk level	CSF present	LTS present	Fish present	CSF adults	CSF larvae	LTS adults	LTS larvae
Big Harrington #1	Control	no	no	no	0	0	0	0
Big Harrington #6	Control	no	no	no	0	0	0	0
Bilk	Control	yes	no	no	79	1,853	0	0
Elk	Control	yes	yes	no	286	286	0	8
Goat	Control	yes	yes	no	142	760	0	14
Mud	Control	yes	yes	no	47	6	0	57
Section 27	Control	yes	no	no	2	0	0	0
Fox Peak, Lower	Low	yes	yes	no	10	0	0	35
Fox Peak, Upper	Low	yes	yes	no	3	292	0	6
Isaac	Low	yes	no	yes	195	1,460	0	0
Isaac Creek	Low	yes	yes	no	19	6	0	10
Running	Moderate	no	no	yes	0	0	0	0
Section 26, Lower	Moderate	no	no	no	0	0	0	0
Section 26, Upper	Moderate	no	no	no	0	0	0	0
Lottie	Elevated	yes	no	yes	77	1	0	0
Lottie, Upper	Elevated	yes	no	yes	100	3	0	0
Maude, East	Elevated	yes	no	yes	57	2	0	0
Maude, North	Elevated	yes	yes	no	62	30	0	7
Maude, West	Elevated	yes	no	yes	121	24	0	0
Stillman	Elevated	yes	no	yes	15	26	0	0

Table 30. Summary of data from gill-net surveys of high mountain lakes in the Clearwater Region, Idaho, in 2018.

Lake	Risk level	Species	Effort (hours)	*n*	CPUE (fish/h)	Mean length (mm)	Mean weight (g)
Isaac	Low	WCT	16	33	2.0	220	118
Running	Moderate	BKT	14	73	5.4	164	43
Lottie	Elevated	BKT	12	19	1.5	186	123
Lottie, Upper	Elevated	BKT	15	5	0.3	157	95
Maude, East	Elevated	WCT	16	8	0.5	267	218
Maude, West	Elevated	WCT	16	4	0.3	309	329
Stillman	Elevated	WCT	19	2	0.1	212	89

BKT = Brook Trout; WCT = Westlope Cutthroat Trout

Table 31. Catch-per-unit-effort (fish/h) for all gill net surveys conducted in Clearwater Region, Idaho, high mountain lakes surveyed in 2018.

		Survey number				
Lake	Risk level	1	2	3	4	5
Isaac	Low	5.6	5.1	2.0		
Running	Moderate	5.9	4.1	5.4		
Lottie	Elevated	4.1	3.4	0.5	1.5	
Lottie (upper)	Elevated	0.7	2.1	0.5	0.3	
Maude West	Elevated	0.9	1.1	0.4	0.3	
Maude East	Elevated	1.0	1.9	0.4	0.5	
Stillman	Elevated	1.4	3.2	0.4	0.8	0.1

Table 32. Variables and associated p-values for generalized linear model analysis of Columbia spotted frog and long-toed salamander presence in high mountain lakes surveyed in the Clearwater Region, Idaho, in 2018. Non-significant values were removed stepwise in the order listed below, with significance set to α = 0.10.

Columbia spotted frog

Variable	Coefficient	Std. error	z	P
Fish presence	0.349	1.968	0.177	0.847
Elevation	-0.095	0.048	-1.983	0.724
Max depth	0.122	0.068	1.796	0.683
% emergent vegetation	0.189	1.249	0.752	0.229
Fines	0.457	0.277	0.821	0.197
% littoral zone	-0.073	0.672	-1.077	0.154
Julian Date	0.766	2.176	0.034	0.017
Julian Date 2	-0.095	0.884	-0.095	0.013

Long-toed salamander

Variable	Coefficient	Std. error	z	P
Fines	0.001	0.012	0.779	0.821
Max depth	0.630	7.861	0.620	0.622
% littoral zone	-0.467	3.752	-1.125	0.538
Elevation	0.013	0.081	1.065	0.507
% emergent vegetation	0.402	1.940	0.107	0.310
Fish presence	-2.742	7.093	-4.146	< 0.001
Julian Date	0.225	0.338	0.566	< 0.001
Julian Date 2	-0.001	0.001	-0.695	< 0.001

Table 33. Summary of historic water conditions compared to Columbia spotted frog (CSF) and long-toed salamander (LTS) presence in high mountain lakes of the Clearwater Region, ID, that were dry during surveys conducted in 2018.

		Survey Year														
		2008			2012			2013			2015			2018		
Lake	USGS	Water	CSF	LTS	Water	CSF	LTS	Water	CSF	LTS	Water	CSF	LTS	Water	CSF	LTS
Section 27	intermittent	-	-	-	Y	Y	Y	-	-	-	low	N	Y	low	Y	N
Section 26 (upper)	perennial	Y	N	Y	Y	Y	N	-	-	-	dry	N	N	dry	N	N
Section 26 (lower)	perennial	Y	N	N	Y	N	N	-	-	-	dry	N	N	dry	N	N
Big Harrington #1	marsh	-	-	-	-	-	-	Y	N	N	dry	N	N	dry	N	N
Big Harrington #6	marsh	-	-	-	-	-	-	dry	N	N	dry	N	N	dry	N	N

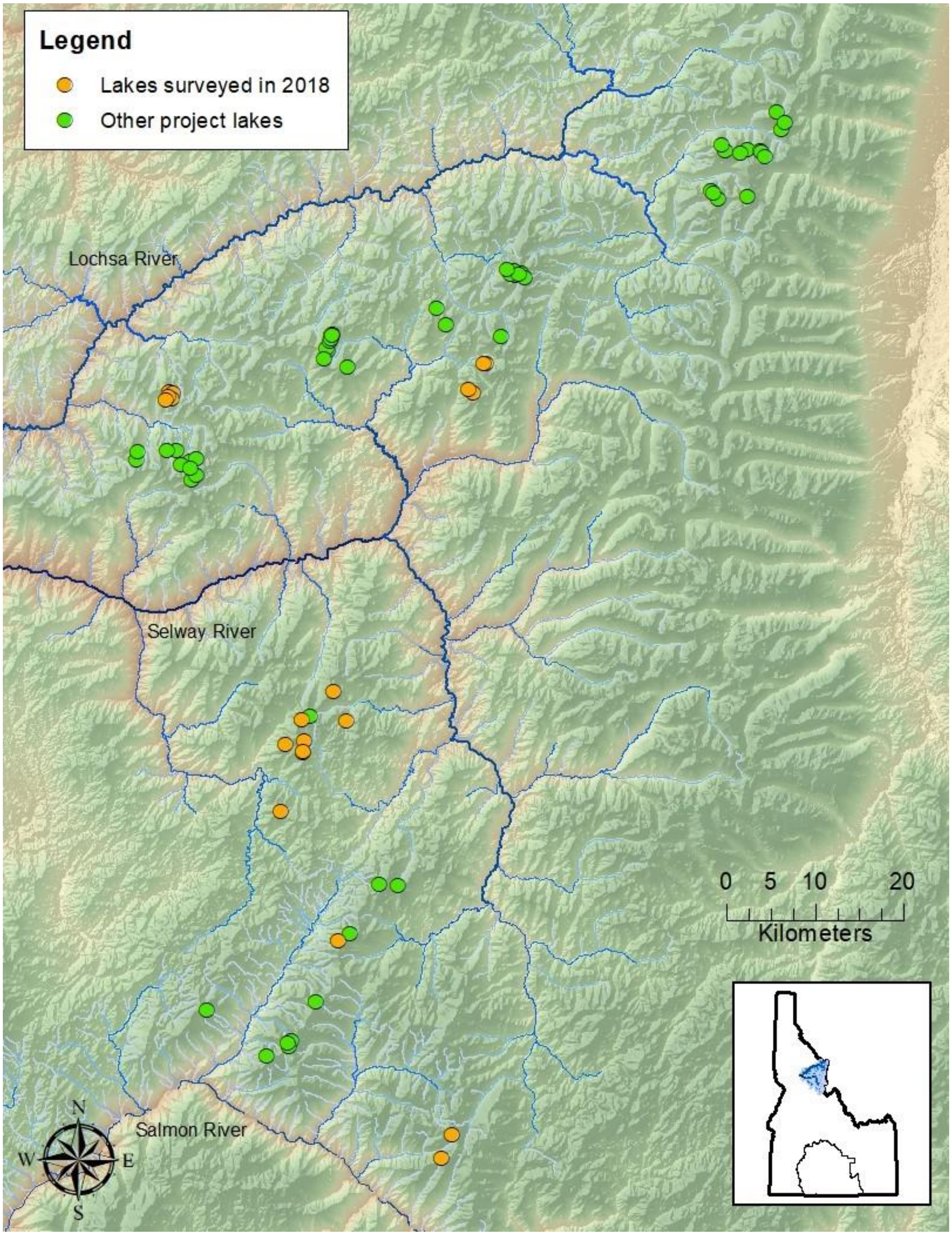

Figure 56. Map depicting high mountain lakes selected to evaluate long term trends in amphibian populations in the Clearwater Region of Idaho, including those lakes surveyed during 2018.

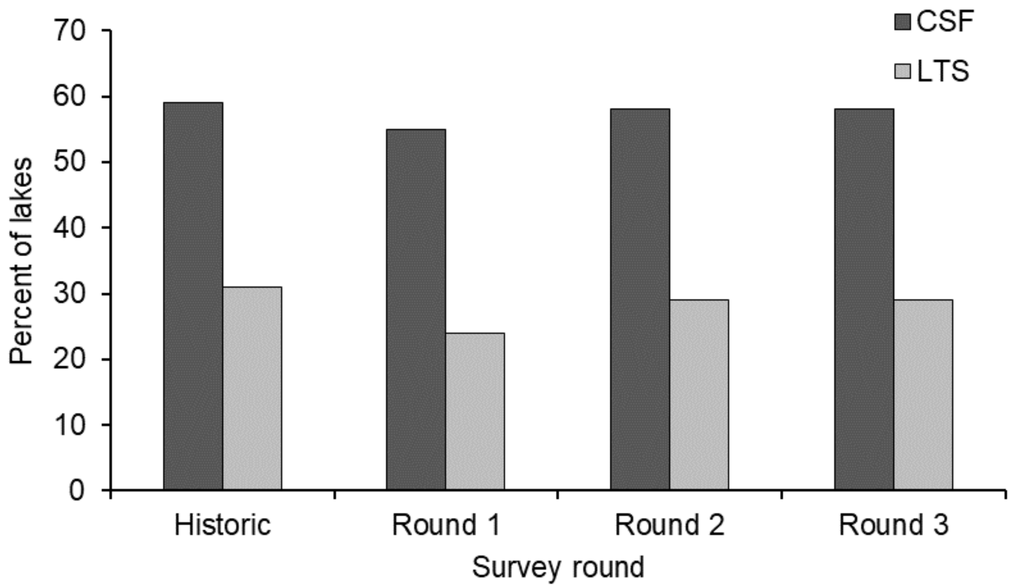

Figure 57. Presence of Columbia spotted frog (CSF) and long-toed salamander (LTS) in high mountain lakes of the Clearwater Region, Idaho, which have historic data and three rounds of surveys completed (n = 63).

Figure 58. Photos showing surface elevation changes in Section 26 Lakes (Upper and Lower), Idaho, from 2013 (top) to 2018 (bottom).

Figure 59. Photos showing surface elevation changes in Big Harrington Lake #1, Idaho, from 2013 (left) to 2018 (right).

Figure 60. Photos showing surface elevation changes in Section 27 Lake, Idaho, from 2012 (left) to 2018 (right).

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